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Aquatic eutrophication indicators in LCA: Methodological challenges illustrated using a case study in New Zealand

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ABSTRACT

Objective: The objective of this work was to determine the effects and implications of generic and site-specific aquatic eutrophication potential indicators in the Life Cycle Assessment (LCA) of livestock farm systems using a New Zealand (NZ) lake catchment case study.

Method: Average dairy and sheep & beef farm systems in the Lake Taupo catchment were studied. Emissions of nitrogen (N) and phosphorus (P) to waterways, and ammonia and nitrogen oxides to air from these farms were calculated using the site-specific OVERSEER® nutrient budget model. These emissions data were then used to calculate the increase in nutrients in water bodies and aquatic Eutrophication indicators with a range of Life Cycle Impact Assessment (LCIA) methods.

Results: Eutrophication indicator results varied considerably depending on the environmental mechanisms modelled by the LCIA method for the fate of N and P, accentuated by different choices for the inventory modelling. Using default emission factors instead of site-specific ones overestimated eutrophication impact results. The most recent methods are not only spatially-explicit and applicable beyond Europe, but they also account for more environmental mechanisms for the fate of the nutrients, giving relatively lower calculated impact results. However, the appropriate scale and spatial resolution is still a crucial question to address for these methods since they greatly affect results. Regarding eutrophication damage assessment, when the actual background nutrient concentration is very low, the end-point assessment method for freshwater eutrophication is not applicable. In this case, LCA fails to account for a high standard of water quality that is in a near-pristine state, but deteriorating.

Conclusions: The inventory of nutrient flows at a farm scale and fate factors modelled at a catchment scale should be site-specific. Freshwater eutrophication indicators should be based on a site-specific (and globally-valid) LCIA model rather than a generic one. Currently-accepted freshwater eutrophication indicators focus only on P, thus capturing only part of the problem for freshwater bodies that are co-limited by N and P (in terms of algal growth) such as Lake Taupo in NZ. Lake Taupo water quality concerns and regulations are not focused on P, but solely on N due to increasing N levels over time. Conclusions from this study are valid beyond NZ and beyond agricultural systems. Future work needs to investigate coupling N and P fate modelling based on the most recent globally-valid and spatially-explicit LCIA methods.

1. Introduction

Aquatic eutrophication is a major water quality issue throughout the world (Khan and Mohammad, 2014). Eutrophication covers all impacts of excessively high environmental levels of macronutrients, the most important of which are nitrogen (N) and phosphorus (P) (Guinée et al., 2002). Eutrophication has many negative effects on aquatic ecosystem, with the most obvious one being the increased growth of algae (Carpenter et al., 1998). N and P are generally the main limiting nutrients that control carbon fixation and therefore growth of plant biomass (e.g. algae) in waterways (Kitsiou and Karydis, 2011). The theory of the limiting nutrient for algal growth originate from the application of the Redfield molar ratio of 1:16 for P to N for optimal algal growth (Redfield, 1958), this assumes that if more than 16 mol of N are present for every mole of P, growth is limited by the amount of P available (P-limited).

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In New Zealand (NZ), river water quality is a matter of concern to the public (Larned et al., 2016). The National Policy Statement for Freshwater Management sets out the objectives and policies for freshwater management, aiming to protect the quality of waterways (MfE, 2014). Minimum acceptable values for N and P concentrations in waterways were defined, and have to be reached within a reasonable timeframe. In this context, assessing the contribution of NZ livestock farms to aquatic eutrophication is pertinent, since they are the main anthropogenic source of nutrients in NZ water bodies (Scarsbrook and Melland, 2015).

Various Life Cycle Impact Assessment (LCIA) methods exist to characterise eutrophication impacts, but they differ in terms of their inventory requirements, geographical coverage, spatial resolution, and emission pathways modelled. Although the eutrophication cause-and-effect chain is modelled with more or less details according to methods, assessing eutrophication potential impact relies successively on: 1. The inventory of nutrients emitted in the environment 2. The fate of these nutrients (Fate Factors, FF), representing their transport and attenuation (e.g. denitrification) from the source of emission to the final compartment (freshwater or coastal marine water). This allows the calculation of a midpoint impact indicator representing a nutrient increase in water. 3. The ecosystem exposure to nutrient enrichment (Exposure Factor) and the effect (Effect Factor, EF) on aquatic species. This allows the calculation of an endpoint indicator representing eutrophication damages on ecosystems. LCIA methods addressing aquatic eutrophication have evolved over time and the methods addressed in this paper are briefly presented chronologically below. Eutrophication impacts calculated with the CML method (Heijungs et al., 1992) assess both terrestrial and aquatic eutrophication in a single indicator, where all emissions (N and P to air, water, soil, and organic matter as COD to water) are aggregated using the Redfield ratio which provides characterisation factors (Redfield, 1958). Thus, the characterisation factors (or "equivalency factors") is independent of whatever substances happen to be the limiting factor for algae growth in a given location (Guinée et al., 2002). This method assumes that 100% of the emissions will contribute to eutrophication, meaning that the fate (transport and attenuation) of the nutrient is not modelled. As a result, CML corresponds to a "worst case scenario" since it ignores that only a fraction of the emissions will be transported to the aquatic environment (Struijs et al., 2009).

Eutrophication impacts calculated with ReCiPe 2008 (Struijs et al., 2009) assess aquatic eutrophication through two distinct impact indicators; marine eutrophication and freshwater eutrophication. This method was recommended by the European Commission (JRC-IES, 2011), notably because it accounts for the sensitivity of the receiving water body: marine water is considered to be N-limited (i.e.: N is the limiting nutrient for marine biomass growth), whereas freshwater is considered to be P-limited (i.e.: P is the limiting nutrient for freshwater biomass growth) (Struijs et al., 2009). In ReCiPe, the Fate Factors (FF) for N and P to marine and freshwaters are site generic, but are derived from a European model (CARMEN & EUTREND) which make them specific to Europe.

Eutrophication impacts calculated with more recent methods account for the fate of N and P using two distinct models. Freshwater eutrophication FF (for P) was developed by Helmes et al. (2012), representing the persistence of P in the freshwaters, and marine eutrophication FF (for N) was developed by Cosme et al. (2017b), representing the persistence of the N exported in the receiving coastal Large Marine Ecosystems. One major improvement is that they both have a global coverage while being spatially-explicit at the country (Helmes et al., 2012), catchment (Cosme et al., 2017b) or grid cell scale (Helmes et al., 2012). The Helmes et al. (2012) method is implemented in ReCiPe 2016 (Huijbregts et al., 2016) and IMPACT World + (Hernandez-Padilla et al., 2017). In their assessment of the maturity of newly-developed versus established LCIA methods, Bach and Finkbeiner (2017) showed that methods with specific and sophisticated models of the impact pathway are covering less contributing substances.

A review of LCIA methods applied to animal production systems showed that CML and ReCiPe 2008 are used regardless of the geographical location of the systems studied. For example, Salou et al. (2017) used CML for milk production in France, whereas Santos et al. (2017) used ReCiPe 2008 outside Europe for cheese production in Brazil. To our knowledge, there are no published applications of Helmes et al. (2012) and/or Cosme et al. (2017b) methods to animal production systems. The wide range of eutrophication impact results reported in LCA studies is due to different approaches for estimating nutrients emitted to water and variation in the contributing substances accounted for, as shown by Costello et al. (2015) for milk in the USA. Thus, there is a need to harmonise LCA approaches. This is reflected by many international initiatives aiming at providing global guidance such as LEAP (LEAP, 2017) and the UNEP/SETAC Life Cycle Initiative (Frischknecht et al., 2016). Nevertheless, there is a lack of studies comparing available eutrophication indicators, a necessary requirement toward harmonization. This paper contributes to the assessment of available LCIA models for eutrophication.

The objective of this work was to compare and evaluate different LCIA methods and their relevance to estimating aquatic eutrophication impacts using NZ's largest lake, Lake Taupo, as a case study. Using case study farms, we calculated aquatic eutrophication indicators with different methods presented in the Material and methods section. Then, in the Result and discussion section, we determined the implications of using generic and site-specific eutrophication potential indicators in the LCA of livestock farm systems at the inventory, midpoint and endpoint impact assessment level. For each LCIA method, we evaluated the modelling of N and P nutrients from emission to damages on the environment, in terms of attenuation processes accounted for, site-specificity and geographical validity.

2. Material and methods

Emissions and impacts were calculated for the farm stage only, on a per-hectare basis for one year.

2.1. Case study farms

The livestock farm systems studied were an average dairy farm and an average sheep & beef farm from the Lake Taupo catchment (volcanic soil with rainfall of 1300 mm/year) (Thorrold and Betteridge, 2006). All farms have livestock grazing perennial grass/clover pastures all year round. The 100 ha dairy farm has 270 cows, uses no brought-in feed and applies fertilisers at a rate of 100 kg N/ha/year and 46 kg P/ha/year. The 480 ha sheep & beef farm is stocked at 11.5 sheep-equivalents/ha (sheep-equivalent represents the amount of feed required for one sheep and lamb), with a 70:30 sheep:cattle ratio and applies fertilisers at a rate of 17 kg N/ha/year and 22 kg P/ha/year.

2.2. Inventory of nutrient flows

Based on primary data for inputs on farm, field emissions of N leaching and P runoff were calculated using the OVERSEER® nutrient budget model hereafter called OVERSEER (Wheeler et al., 2007), and for ammonia and nitrogen oxides using NZ-specific emissions factors from the NZ Greenhouse Gas Inventory (MfE, 2015). Nitrogen oxides volatilised from fertiliser and excreta are negligible in comparison to ammonia (Sherlock et al., 2008), so volatilised gas was considered 100% ammonia. OVERSEER is a nutrient model which has been validated against field site measurements from throughout NZ (McDowell et al., 2005; Wheeler et al., 2007). Nitrogen leaching is estimated based on amount and timing of N excreted by animals (urine, dung and farm dairy effluent) and fertilisers applied, and is mainly driven by drainage volume and soil properties (Wheeler et al., 2011). An attenuation factor of 50% was used to estimate the fate of N from soil (below root-zone) to freshwaters (Elliot et al., 2014) when applying methods that do not account for it (i.e. CML and ReCipe 2008 when using FF based on net emissions, refer to Fig. 1). Phosphorus runoff is estimated based on soil, climate, hydrologic conditions and management factors (application rates and transport factors) (McDowell et al., 2005). OVERSEER is not only a tool for farm management, and for estimating site-specific field emissions for LCA (Chobtang et al., 2017; Zonderland-Thomassen et al., 2014; Basset-Mens et al., 2009), but also for policy on freshwater management. Indeed, this model is used to define maximum N leaching limits for farms in the Lake Taupo catchment (Ledgard et al., 2009).

2.3. Eutrophication impact assessment

Eutrophication impacts were calculated with CML, ReCipe 2008, ReCiPe 2016 (using Helmes et al., 2012) and Cosme and colleagues' (Cosme et al., 2015, 2017a; Cosme and Hauschild, 2016; Cosme et al., 2017a; Cosme and Hauschild, 2017; Cosme et al., 2017b) models. We distinguished three stages in the impact assessment models; (i) the increase in nutrients in the receiving water body (N and P fate modelling) (section 2.3.1.), (ii) this increase divided by a reference emission in order to obtain a eutrophication indicator (midpoint) (Huijbregts et al., 2016; Struijs et al., 2009) (section 2.3.2.), or (iii) multiplied by an Effect Factor (EF) (and sometimes an exposure factor) as an indicator for ecosystem damage (endpoint) (Cosme et al., 2017a) (section 2.3.3.).

2.3.1. Nutrient fate modelling

To calculate an increase in nutrients in a water body, each method relies on different inventory requirements and different FF. The inventory can be based on gross supply (e.g. fertiliser applied on agricultural land) or on net emissions to the environment (e.g. phosphate runoff into rivers). Then the FF links the quantity released into the environment to the quantity eventually reaching a given compartment (Rosenbaum et al., 2007) (freshwater or marine water), accounting for transport and attenuation with an accuracy that varies a lot between methods. CML is based on net emissions of N and P to freshwater and assumes 100% of emissions will contribute to eutrophication (pathway A in Fig. 1). Conversely, impacts calculated with ReCiPe 2008 and ReCiPe 2016 can be derived using either gross supply of fertilisers and manure to agricultural soil, or net emissions to water. Thus there are two options with ReCiPe, using either the $FF_{gross\ supply}$ which accounts for the fate from agricultural topsoil to final receiving water compartment (freshwater or sea) (pathway B in Fig. 1), or using the $FF_{net\ emission}$ which only accounts for the fate of nutrients from freshwater to final receiving water compartment (pathway C in Fig. 1). In this study, we have used both. We compared ReCiPe 2008 and ReCiPe 2016 using impacts assessed based on gross supply (B and E in Fig. 1) or based on net emissions that account for our site-specific nutrient emissions modelling (C and D in Fig. 1). Note that in pathway E, we applied Helmes et al. (2012) FF using the ReCiPe 2016 assumption of 10% P losses from agricultural soil (Bouwman et al., 2009). In pathway C and D, we multiplied the net emissions

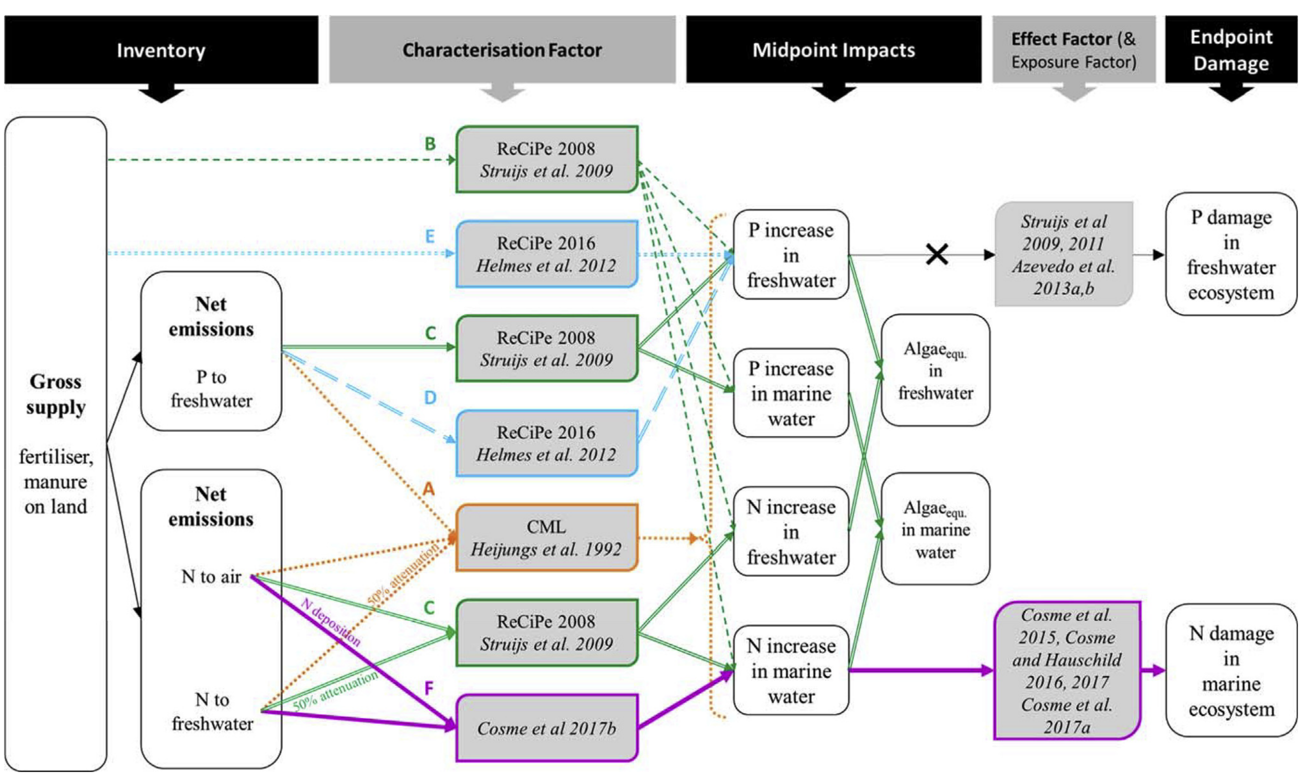

Fig. 1. Life Cycle Impact Assessment (LCIA) methods applied in this study illustrated throughout the eutrophication cause-and-effect chain in LCA. Nutrient inventory flow requirements (gross supply or net emissions), Characterisation Factors (CF = Fate Factor/Fate Factor $_{reference}$, except for CML where CF = equivalency factor), midpoint impacts indicators, exposure and effect factors and endpoint damage indicators vary with methods. Capital letters (A–F) refer to pathways and methods applied in this study.

Table 1
Fate Factors (FF) and Characterisation factors (CF) applied for each method. ReCiPe 2008 CF has to be multiplied by the volume of marine or freshwater to obtain a dimension of year (See Section 3.2). DIN = dissolved inorganic nitrogen, *n.a* = not available.

	Method	Substance emitted	Compartment	Region (scale)	FF	FF unit	CF	CF unit/kg substance emitted
N	CML (Heijungs et al., 1992)	Nitrogen compounds	Air, Water, Soil	*n.a*	*n.a*	*n.a*	0.42	kg PO_4^{3-}eq
	ReCiPe 2008 (Struijs et al., 2009)	Fertiliser applied (N)	Soil + Air	*n.a*	5.21E-06	yr/km^3	0.073	kg N_{eq}
		Manure applied (N)	Soil + Air	*n.a*	5.69E-06	yr/km^3	0.079	kg N_{eq}
		Nitrogen *(reference)*	Water	*n.a*	7.17E-05	yr/km^3	1	kg N_{eq}
		Ammonia	Air	*n.a*	6.60E-06	yr.tN/km^3.tNH_3	0.092	kg N_{eq}
	Cosme et al., 2017b	Nitrogen (DIN)	Water (river)	Waikato (NZ region)	33.5	days	0.370	kg N_{eq}
		Nitrogen (DIN) *(reference)*	Water (river)	World average	90.5	days	1	kg N_{eq}
	using ReCiPe 2008 freshwater FF_{NH3}	Ammonia	Air	n.a	5.93	days	0.065	kg N_{eq}
P	CML (Heijungs et al., 1992)	Phosphorus compounds	Air, Water, Soil	*n.a*	*n.a*	*n.a*	3.06	kg $PO_4^{3-}{}_{eq}$
	ReCiPe 2008 (Struijs et al., 2009)	Fertiliser applied (P)	Soil	*n.a*	1.83E-05	yr/km^3	0.053	kg P_{eq}
		Manure applied (P)	Soil	*n.a*	1.72E-05	yr/km^3	0.05	kg P_{eq}
		Phosphorus *(reference)*	Water	*n.a*	3.44E-04	yr/km^3	1	kg P_{eq}
	ReCiPe 2016 (Helmes et al., 2012 FFs)	Phosphorus	Soil	NZ (country)	0.62	days	0.005	kg P_{eq}
		Phosphorus	Freshwater	NZ (country)	6.2	days	0.05	kg P_{eq}
		Phosphorus	Freshwater	From Taupo to marine water	160.7	days	1.236	kg P_{eq}
		Phosphorus	Freshwater	Taupo grid cell	9.6	days	0.074	kg P_{eq}
		Phosphorus *(reference)*	Freshwater	World average	85.0	days	1	kg P_{eq}

of N and P to freshwater and to air (ammonia and nitrous oxides), by their appropriate FF $_{\text{net emission}}$. Finally, the Cosme et al. (2017b) method, is based on net emissions and provides FF for waterborne N emissions only (pathway F in Fig. 1). Since the methods do not currently include atmospheric N deposition, we accounted for N deposition from ammonia and nitrogen oxides to be more complete and used the ammonia FF from ReCiPe 2008. The volume of NZ Large Marine Ecosystem (LME) was used to convert FF units from year/km^3 to years, based on Cosme and Hauschild (2017). All FF are summarised in Table 1.

2.3.2. Eutrophication midpoint impacts

With CML, eutrophication (terrestrial and aquatic) is calculated based on a Characterisation Factor (CF) to convert all nutrient flows to phosphate equivalents ($PO_4^{3-}{}_{eq}$) (Heijungs et al., 1992). With ReCiPe 2008, a CF is calculated as the ratio of the FF and a reference emission ($CF=FF/FF_{\text{reference emission}}$) which is P emitted in freshwater for freshwater eutrophication or N emitted in freshwater for marine water eutrophication (Cf. Table 1). The approach is slightly different with ReCiPe 2016 and Cosme et al. (2017b) where CFs are calculated as the ratio of the FF and the world average FF ($CF=FF/FF_{\text{world average}}$) which are 85 days for P to freshwater (ReCiPe 2016) and 90.5 days for N to freshwater (Cosme et al., 2017b) (Table 1). Note that since there is a discrepancy in CFs between ReCiPe 2016 and Helmes et al. (2012), we reverted to using CFs from the original publication by Helmes et al. (2012). ReCiPe 2016 does not address marine eutrophication because there is not an appropriate endpoint model (Huijbregts et al., 2016), thus it does not align with recent work from Cosme and colleagues (Cosme et al., 2015, 2017a; Cosme and Hauschild, 2016; Cosme et al., 2017a; Cosme and Hauschild, 2017). All CF are summarised in Table 1.

We also calculated an alternative midpoint impact indicator to consider both N and P nutrients in a given water compartment (marine or freshwater). To do so, we aggregated N and P nutrients in each receiving compartment using conversion factors for P and N in terms of algae, based on the Redfield Ratio (Redfield, 1958 used by Goedkoop et al., 2009). Results are expressed in kg algae-eq/ha.

2.3.3. Eutrophication endpoint damage

The eutrophication endpoint refers to damage of ecosystems from a nutrient increase. The CML method does not assess damages and therefore is only considered a midpoint methodology.

ReCiPe 2008 and ReCiPe 2016 methods evaluate the effects of P enrichment on freshwater ecosystems only. ReCiPe 2008 EF is based on a stressor-response relationship representing the effects of P concentration in Dutch freshwater ecosystems (Struijs et al., 2011). ReCiPe 2016 EF is based on an improved effects modelling accounting for more species and freshwater types (Azevedo et al., 2013a, b) than Struijs et al. (2011). The Azevedo et al. (2013a, b) effect model is based on a richer data set of P effects on ecosystems throughout the world and at a lower taxonomic level (species instead of genus level).

Regarding N effects on marine ecosystems, Cosme and Hauschild (2017) modelled mechanistically an ecosystem exposure model for dissolved inorganic nitrogen absorbed by primary producers in coastal waters and the biological processes that result in oxygen depletion (for each Large Marine Ecosystem on the planet), and an effect model based on sensitivity of benthic species (in the depth of the ocean) to oxygen depletion (for 5 climatic zones). We applied the exposure factor (quantifying the oxygen consumption due to N enrichment (Cosme et al., 2015)), and the EF (assessing the effects of oxygen depletion on benthic ecosystems (Cosme and Hauschild, 2016)), and ultimately calculated endpoint and damage to ecosystems (Cosme et al., 2017a)).

3. Results and discussion

3.1. Inventory of nutrient flows

Table 2 shows the inventory of nutrient flows for the sheep & beef and dairy farms per-hectare per-year. N leaching and P runoff are respectively 2.9 times and 2.7 times higher for dairy farms, but dairying land only represents less than 3% of the pastoral land area of the Lake Taupo catchment (most is in sheep & beef farming; Vant and Huser (2000)). Thus, converting sheep & beef pasture to dairying would increase the overall N load to the lake by 20–60% (Vant and Huser, 2000).

Default emission factors are not appropriate where field-specific estimates of emissions are available, as for the Lake Taupo catchment. Regarding N, default volatilisation rates in ReCiPe 2008 are

Table 2
Inventory results for nutrient flows [kg N or P/ha/year] and emission compartments. Nutrient flows are separated in inputs, internal cycling (within the system studied) and outputs.

	Nutrient flow	Compartment	Sheep & Beef	Dairy	Source
Inputs	N fertiliser	agri. soil	17.0	103.0	Primary data
	N fixation (clover) and atmospheric deposition	agri. soil	62.0	125.0	OVERSEER
	P fertiliser	agri. soil	22.0	45.0	Primary data
Internal cycling flows	N excreta (dung & urine)	agri. soil	162.0	370.0	OVERSEER
	N in farm dairy effluent (FDE)	agri. soil	–	19.5	OVERSEER
	P in excreta	agri. soil	15.1	30.6	OVERSEER
Outputs	N out in wool, meat, milk		19.0	62.0	OVERSEER
	N leaching (below root-zone)	soil	16.0	47.0	OVERSEER
	NH_3-N and NO_x volatilisation	air	17.9	49.2	(MfE, 2015)
	N_2O-N (direct & indirect)	air	1.6	4.5	(MfE, 2015)
	P out in wool, meat, milk		3.0	11.0	OVERSEER
	P runoff	freshwater	1.1	3.0	OVERSEER

21% of N in manure and 7% in fertiliser, whereas NZ-specific volatilisation rates were 10% of N in manure and fertiliser (MfE 2015). For N leaching, using default IPCC emission factors instead of OVERSEER's outputs would have greatly overestimated N emissions to water by 3.4 times and 3.0 times for sheep & beef and dairy respectively. Regarding P, the default transfer-fraction from soil to water in ReCiPe 2016 is 10%, whereas the site-specific fraction estimated via OVERSEER is smaller at 3% for sheep & beef and 4% for dairy. Again, using default factors would have overestimated P emissions to water.

Differences in terms of technosphere and ecosphere boundary across LCIA methods are confusing for the LCA practitioner, since they rely on different inventory requirements. Furthermore, there is no agreement on emission models to be used for the inventory. Regarding emissions from fertilisers, a consensus is still missing on a globally applicable model that calculates soil and water emissions under specific soil conditions (e.g. pH, clay content, slope, etc.) (Notarnicola et al., 2017). Since there is no consensus, we applied in this study a nationally-accepted, published and validated model (OVERSEER, Wheeler et al. (2007)). There is a need for a guideline for good practices in calculating emissions for the inventory analysis, which should be helped by the on-going work of the UNEP/SETAC Life Cycle Initiative Task force on Acidification and Eutrophication.

3.2. Eutrophication midpoint impacts

Table 3 shows eutrophication impacts calculated with various methods. Comparison of the results from different methods is not straightforward; not only because the methods address different processes of nutrient fate, but also because the rationale and units of indicators are different. The CML single terrestrial and aquatic eutrophication indicator cannot be compared with freshwater or marine water eutrophication indicators. Freshwater eutrophication results calculated with ReCiPe 2008 were 21 times higher than calculated with ReCiPe 2016, whereas marine eutrophication results calculated with ReCiPe 2008 were 1.2 times higher than calculated with Cosme et al. (2017b) (Table 3). In the following sections, we analyse in detail the differences between methods by comparing the fate of N and P from soil to the final compartment.

3.2.1. Phosphorus

Impact Results based on gross vs. net emissions – We compared freshwater eutrophication estimates based on gross supply of fertilizer and manure (B and E in Fig. 1) versus net emissions to air and freshwater (C and D in Fig. 1). With ReCiPe 2008, the estimates were higher when based on gross supply compared to net emissions (Table 4), by 1.7 fold for sheep & beef and by 1.3 fold for dairy. This was due to the CARMEN model only accounting for plant uptake and topsoil binding, but not for other P transport or attenuation processes, whereas OVERSEER accounts for accumulated P in soil attached to sediments and lost via P runoff and erosion. With ReCiPe 2016, the difference between impacts calculated with gross vs. net emissions was even bigger. Freshwater eutrophication potential was 2.5–3.4 times higher when based on gross P emissions instead of net P emissions to water estimated with OVERSEER. This shows that the default assumption (from ReCiPe 2016) of 10% of P inputs being lost from agricultural soil to surface water is not appropriate for this system. This fraction of P loss varied across other LCIA methods, with net emission FFs being 7 to 20 times higher than gross supply FFs (Potting et al., 2005; Huijbregts and Seppälä, 2001). This illustrates the preference for use of spatially-explicit fate models, accounting notably for site-dependent P concentration in soil, which is one of the most important parameters influencing P emissions to water from agriculture (Scherer and Pfister, 2015). In the following paragraph, we present results calculated using net emissions.

P fate in freshwater - We also compared the P increase in freshwater estimated with ReCiPe 2008 with ReCiPe 2016 (Table 5). To allow a comparison, ReCiPe 2008 FFs had to be multiplied by the total volume of European freshwater (885 km^3, according to Struijs et al., 2009), to convert a dimension of concentration (kg P/km^3) to a dimension of mass (kg P). Results showed that the P fate estimate using ReCiPe 2008 was 17.9 times higher than that using ReCiPe 2016. The FF for P emissions to freshwater was much lower with ReCiPe 2016 (Helmes et al., 2012) (6.2 days for NZ) than with ReCiPe 2008 (Struijs et al., 2009) (111 day for Europe, used for NZ). This is because P fate modelling in the CARMEN model only accounts for the advective transport of P and does not include the P removal processes through retention (uptake by biomass and adsorption to suspended solids in waterways) and water withdrawal, as modeled by Helmes et al. (2012.

Helmes et al. *(2012)* – The Helmes et al. (2012) model was identified as being most relevant in describing the environmental mechanism of the cause-and-effect chain for P (the “environmental relevance” criteria was defined by JRC-IES (2011) to evaluate LCIA methods). However, it is important to note that not all environmental mechanisms are modelled. Particulate P is not yet accounted for in the transfer fraction from agricultural soil to freshwater, but P losses in particulate form (related to erosion) are significant in NZ (Cournane et al., 2011).

3.2.2. Nitrogen

Impact Results based on gross vs. net emissions – Marine eutrophication results were higher when based on gross supply of fertilizer and manure compared to net emissions with ReCiPe 2008 (Table 4). This was due to the leaching fraction estimated with OVERSEER combined with the 50% attenuation factor from root

Table 3
Comparison of eutrophication impact indicator results (per ha) from an average dairy farm and an average sheep and beef farm in the Lake Taupo catchment, calculated with CML, ReCiPe 2008, ReCiPe 2016 and Cosme et al. (2017b), all based on net emissions estimated with OVERSEER. For the sake of comparison with other methods, CML impact results in kg P_{eq} are: 4.7 for sheep & beef and 13.1 for dairy.

Pathway in Fig. 1	Method	Impact indicator	Unit	Compartment	Sheep& Beef	Dairy
A	CML (Heijungs et al., 1992)	Eutrophication	kg PO_4^{3-} eq	Terrestrial & aquatic	14.4	40.0
C	ReCiPe 2008 (Struijs et al., 2009)	Marine eutrophication	kg N_{eq}	Marine water	10.0	29.0
		Freshwater eutrophication	kg P_{eq}	Freshwater	1.10	3.00
D	ReCiPe 2016 (Helmes et al., 2012)	Freshwater eutrophication (Based on Country scale CF)	kg P_{eq}	Freshwater	0.05	0.14
F	Cosme et al. (2017b)	Marine eutrophication	kg N_{eq}	Marine water	8.5	24.6

Table 4
Comparison of marine or freshwater eutrophication impact indicator results (per ha) based on gross versus net emissions, calculated with ReCiPe 2008 and ReCiPe 2016.

Pathway in Fig. 1	Method	Eutrophication	Comment	Unit	Sheep & Beef	Dairy
B	ReCiPe 2008 (Struijs et al., 2009)	Marine	based on gross supply of fertilisers & manure to soil	kg N_{eq}	14.0	38.3
C		Marine	based on net emissions to air and water	kg N_{eq}	10.0	29.0
B		Freshwater	based on gross supply of fertilisers & manure to soil	kg P_{eq}	1.9	3.9
C		Freshwater	based on net emissions to air and water	kg P_{eq}	1.1	3.0
E	ReCiPe 2016 (Helmes et al., 2012)	Freshwater	based on gross supply of P fertiliser & manure to soil and CF at country scale	kg P_{eq}	0.18	0.36
D		Freshwater	based on net P emissions to water and CF at country scale	kg P_{eq}	0.05	0.14

zone to freshwater being less than N estimated to reach freshwaters using the CARMEN model. In the following paragraph, we analyse results calculated with net emissions.

N fate from soil to marine water — We compared the N increase in marine water estimated with ReCiPe 2008 and Cosme et al. (2017b) (Table 5). To allow a comparison, ReCiPe 2008 FF had to be multiplied by the volume of the New Zealand Shelf Large Marine Ecosystems (4490.2 km^3, Cosme et al. (2017a)), to convert a dimension of concentration (kg N/km^3) to a dimension of mass (kg N). Results showed that the N fate estimate using ReCiPe 2008 was 1.5 times higher than that using Cosme et al. (2017b). The FF for N emissions to freshwater is much higher for ReCiPe 2008 (118 days) than for Cosme et al. (2017b) (33.5 days). An analysis of the N fate modelling reveals discrepancies in N loss estimates from soil to river, from river to marine water and eventually in the ocean. Regarding losses of N from soil to river, we assumed a 50% N attenuation when applying ReCiPe 2008, whereas Cosme and colleagues' model estimates a 0% N removal (function of runoff). If we applied a 50% attenuation factor for Cosme and colleagues' model as well, the results would have been even lower. Regarding N losses from river to marine water, the CARMEN model accounts for a constant 30% N removal due to denitrification in freshwaters, whereas Cosme et al. (2017b) accounts for the basin-specific aquatic retention (within river network, within constructed reservoir, and through water abstraction) in the catchment (corresponding to 55% N removal for Waikato river, between Lake Taupo and the sea). Regarding N removal in marine water, this is only accounted for in the method of Cosme and colleagues (through denitrification, sedimentation and advection) (Cosme et al., 2017b).

Site-specific N attenuation - In NZ, there is ongoing research on the characterisation of N attenuation according to the site-hydrogeological specificities. The reported uncertainty for the attenuation factor ranges from 0 to 80% (Elliot et al., 2014). This uncertainty (due to natural variation of denitrification processes) has a great influence on the eutrophication impact result: marine eutrophication assessed with ReCiPe 2008 can vary up to 3.5 fold with the attenuation factor varying from 0 to 80%. Future work for NZ should use site-specific fate modelling of nutrients currently under development in several catchments in NZ (Stenger et al., 2016).

Cosme et al. *(2017b)* — The Cosme et al. (2017b) model was identified as being most relevant in describing the environmental mechanism of the cause-and-effect chain for N, however, it is important to note that not all environmental mechanisms are modelled, and only dissolved inorganic N is accounted for. In this model, dissolved organic N is disregarded although it can potentially act as a source of N, and losses of N from soil to river do not account for denitrification (except for tropical humid areas) although this attenuation occurs in NZ. In the model, the FF is defined at the scale of the large Waikato river catchment, implying

Table 5
Comparison of N and P fate estimates in marine or freshwaters (per ha) calculated with ReCiPe 2008, ReCiPe 2016 and Cosme et al. (2017b). FF= Fate Factor.

Method	Impact indicator	Comment	Unit	Sheep & Beef	Dairy
ReCiPe 2008 (Struijs et al., 2009)	P fate in freshwater	based on the European freshwaters volume from Struijs et al. (2009)	kg P	0.335	0.913
ReCiPe 2016 (Helmes et al., 2012)	P fate in freshwater	based on country-scale FF	kg P	0.019	0.051
ReCiPe 2008 (Struijs et al., 2009)	N fate in marine water	based on the volume of the New Zealand Shelf Large Marine Ecosystems from Cosme et al. (2017a)	kg N	3.22	9.34
Cosme et al. (2017b)	N fate in marine water	based on Waikato FF for N to river and ReCiPe 2008 FF for N to air for freshwater	kg N	2.11	6.09

that FF is identical for emissions located upstream (near Lake Taupo), and for emissions located near the outlet. This is neglecting differences in attenuation for emissions from upstream versus downstream farms.

3.2.3. Geographical validity & spatial scale

There is a need for a globally-valid model, but with site-specific characterisation factors. ReCiPe 2008 is not appropriate for NZ since it is specific to Europe. Nevertheless, in the absence of FFs for other continents, this method has been used outside Europe, such as in Central or South America recently (Huerta et al., 2016; Willers et al., 2017). ReCiPe 2008 is recommended by the European Commission but is not transparent as the modelled fate processes and associated assumptions used in the CARMEN model have not been published (Beusen, 2005). The models developed by Helmes et al. (2012) and Cosme et al. (2017b) are significant improvements because they are applicable globally while being spatially-explicit. With regionalised assessment, applicability is a main issue, and is strongly scale-dependent. The average P FF for NZ showed a standard deviation of 19.6 days (Helmes et al., 2012). When using a FF defined at a half-degree resolution for the Lake Taupo catchment, the P fate estimate is 1.6 times higher than that based on the NZ average (Table 6). This shows that a country average FF may not be appropriate because local hydrological properties have the largest effects on FFs (Helmes et al., 2012). Besides, using a FF for Lake Taupo accounts for impact in the lake, but does not acknowledge that part of the P will continue its journey downstream and have impacts beyond the lake. When using the cumulative FF for the Waikato River catchment instead of the Lake Taupo FF, impact results are 16.7 times higher (Table 6). The appropriate scale thus depends on the objectives of the study. However, there is clearly a trade-off between geographical accuracy and feasibility, which applies for N as well as P. We applied FFs defined at the basin scale in this paper, and that would be time-consuming to do so for the LCA of a global supply chain since regionalisation is not supported by commercial LCA software yet. It is thus paramount that LCA practitioners assess their objectives to determine if site-specificity is required for a given inventory flow and at which scale it should be inventoried. For example, if the objective is the eco-design of farm systems with different location options in NZ, site-specific inventory flows are required, ideally at the catchment scale. However, if it is for a product to a market (e.g. NZ lamb to UK) then nationally-specific inventory flows are appropriate while for some minor inputs with uncertain origin (e.g. pesticides) default inventory flows would be acceptable.

3.2.4. Sensitivity of receiving water bodies

Accounting for the sensitivity of water bodies to eutrophication drivers is important, but doing so by focusing on a single nutrient may be inappropriate. Assuming that marine and freshwater ecosystems have distinct single limiting nutrients, being N and P, respectively, is in contradiction with many demonstrations that freshwater and marine ecosystems can be similar in terms of N and P limitation (Elser et al., 2007). In NZ, freshwaters can be N-limited, P-limited or co-limited (McDowell and Larned, 2008). Lake Taupo is co-limited by N and P (Pearson et al., 2016) (but mostly N limited, which is a typical characteristic of volcanic-derived strata (McDowell et al., 2005)), so the freshwater effect model focused on P is only capturing part of the problem. This applies well beyond NZ: co-limitation was measured for example in Irish rivers (Elsaholi and Kelly-Quinn, 2013), Canadian lakes (Ogbebo et al., 2009), an N-limited freshwater-storage reservoir in Australia (Baldwin et al., 2003), and in lakes in the USA (Jin et al., 2007).

To avoid using the concept of limiting nutrient, we aggregated N and P nutrients in each receiving compartment (marine or freshwater) using conversion factors for P and N in terms of algae. Using this approach, marine water eutrophication was estimated at 92 kg algae$_{-eq}$/ha for sheep & beef and 260 kg algae$_{-eq}$/ha for dairy, and freshwater eutrophication was 81 kg algae$_{-eq}$/ha for sheep & beef and 230 kg algae$_{-eq}$/ha for dairy (calculated based on ReCiPe 2008). This gives an impact indicator that reflects an increase of both N and P in a water compartment.

3.2.5. Perspectives for nutrient fate modelling

Future research work will need to investigate the combination of two fate models for N (Cosme et al., 2017b) and P (Helmes et al., 2012) for co-limited freshwaters. Distinguishing lakes from rivers in the P fate modelling would also be very appropriate (Azevedo et al., 2013b). This will account for more site-specificity for the fate of both nutrients, and within a global model which is applicable to a whole globalised supply chain. It may be important to distinguish on-site eutrophication impacts from off-site/background eutrophication impacts (i.e. rest of the world). Future LCA studies should also include the contribution of background processes (beyond the scope of this paper). Indeed, variation in the accounting of nutrient inputs associated with purchased feed (e.g. fertiliser applied for feed production) causes a large variation in eutrophication results across LCA studies (Costello et al., 2015).

3.3. Eutrophication endpoint damage

3.3.1. Endpoint effects modelling

The concentrations of total N and total P in Lake Taupo are 0.079 mg/L and 0.0052 mg/L respectively, on average between the years 2007–2010 (Vant, 2013). The concentration of P is so low that it is outside the domain of validity of the P effect factor equation, which is stated valid for concentrations above 0.1 mg/L (Struijs et al., 2011) or above 0.05 mg/L (for temperate lakes, according to Azevedo et al., 2013a). As a result, it was not possible to estimate any damages from an increase in P nutrients in Lake Taupo using current methods. We therefore only calculated marine damage from an N increase in the NZ Shelf Large Marine Ecosystem using the Cosme and colleagues' model (Cosme et al., 2015, 2017a; Cosme and Hauschild, 2016; Cosme et al., 2017a). This model successively assesses water oxygen depletion due to N inputs (in kg O_2), the effect of oxygen depletion on marine species richness (in Potentially Affected Fraction of species (PAF).m^3 and Potentially Disappeared Fraction of species (PDF).m^3), and eventually marine eutrophication ecosystem damage using site-dependent species density (Table 7). For the NZ Shelf Large Marine Ecosystem, the damage score is 1.69×10^{-9} species/kg N (Minimum value is 1.6 10^{-12} in the Central Arctic Ocean, maximum value is 4.8×10^{-8} in

Table 6
Freshwater eutrophication impact indicators (per ha) calculated with Helmes et al. (2012) at different spatial scale resolution for the FF. CF was calculated using the world average FF (130 days) as a reference emission. FF= Fate Factor, CF= Characterisation Factor.

FF scale	Unit	Sheep & Beef	Dairy
Based on country-scale FF (NZ average)	kg P_{eq}	0.05	0.14
Based on Taupo cell FF half-degree resolution scale (Taupo)	kg P_{eq}	0.08	0.22
Based on the cumulative FF half-degree resolution scale (Waikato river catchment: From Taupo to the sea)	kg P_{eq}	1.36	3.71

Table 7
Impact indicators (per ha) across the marine eutrophication cause-effect chain: N increase in marine water and marine eutrophication impact (Cosme et al., 2017b; Cosme and Hauschild, 2017), Oxygen consumption due to N inputs (Cosme et al., 2015), End-points and Damages (Cosme and Hauschild, 2016; Cosme et al., 2017a) for NZ Shelf Large Marine Ecosystem.

Impact indicator	Comment	Unit	Sheep & Beef	Dairy
N increase in marine water		kg N	2.1	6.1
Marine eutrophication	using world average N fate in freshwater	kg N_{-eq}	8.5	24.6
Marine water oxygen consumption due to N inputs		kg O_2	12.0	34.6
Marine eutrophication ecosystem response (endpoint)	volume-integrated Potentially Affected Fraction (PAF) of species	PAF.m^3	3342	9627
	volume-integrated Potentially Disappeared Fraction (PDF) of species	PDF.m^3	1671	4813
Marine eutrophication ecosystem damage	damages to ecosystems	species	4.14E-09	1.19E-08

the Northeast U.S. Continental Shelf). Since this method is so recent, there are no other published estimates for comparison with results from Table 7.

3.3.2. LCA endpoint indicators and policy

LCIA endpoint methods consider that freshwater eutrophication damage start with concentrations above an optimum P level for ecosystems, and are consistent with current water quality policies in Europe. Struijs et al. (2011) found the highest number of freshwater species at an average total P concentration of 0.1 mg/L, which is just below the regulatory water quality objectives for European lakes (0.15 mg/L) (European Commission, 2000). The NZ national bottom line is more demanding and is set at 0.05 mg/L for total P in lakes (MfE, 2014). But for Lake Taupo, the objectives of water quality are stricter, with a target to stay below 0.0056 mg/L total P (WRC, 2016). In this case, LCA fails to account for a high standard of water quality that is in a near-pristine state. In addition, by neglecting the effects of N, a freshwater eutrophication indicator focused only on P is in contradiction with local government regulations currently focusing on limiting N inputs due to increasing N concentrations in the lake and associated minor water quality degradation (WRC, 2016). Dairy and sheep & beef farms in the catchment are monitored and have a maximum nitrogen discharge allowance (WRC, 2016), that is determined using OVERSEER (Ledgard et al., 2009), and a water quality objective of 0.070 mg/L total N (WRC, 2016). Thus, quantifying the eutrophication damage of these dairy and sheep & beef farms is pertinent but not feasible with current LCIA methods and assumptions.

There is a need to manage both N and P use to reduce freshwater eutrophication, as highlighted by Conley et al. (2009) and McArthur et al. (2010), because nutrient limitation (N or P or co-limitation) varies with flow and time (McArthur et al., 2010), including from time lag between N losses associated with past and present land use. As a result, the concentration of N may not have reached its maximum, meaning that P-limitation may increase in the future (McDowell et al., 2009; Larned et al., 2016).

3.3.3. Perspectives for effect factor modelling

Future research should focus on several perspectives. First, lakes and rivers should be distinguished in the P effect modelling as demonstrated by Azevedo et al. (2013b). Second, since the current EF for P was not applicable in our case study, should we consider using another indicator for assessing impacts from increased nutrients? ReCiPe 2008 defined aquatic eutrophication as “nutrient enrichment of the aquatic environment” (Struijs et al., 2009). As such, any nutrient enrichment should be considered as contributing to the eutrophication impact. Indeed, a shift from an oligotrophic (low nutrient content) to a mesotrophic state (intermediate nutrient content) should be captured in an impact assessment study, because we are “consuming” our good quality water pool before reaching the damage threshold. LCA should be able to align to environmental policies, such as the regulations enforced in Lake Taupo. For example, other indicators are based on a “distance to target” approach: characterizing the difference between the natural concentration and the targeted concentration (Mekonnen and Hoesktra, 2015).

4. Conclusion

The results of this paper are valid beyond NZ and beyond agricultural systems. The application of eutrophication indicators is thwarted by the inconsistency in the processes accounted for, and a lack of clear guidelines of inventory requirements for LCA practitioners. This paper illustrates the differences between current methods and their limitations. A comparison of eutrophication impact assessment indicators showed a wide variation of results which was accentuated by effects of different choices for the inventory modelling. The inventory of nutrient flows at a farm scale and fate factors modelled at a catchment scale should be site-specific (the appropriate scale is still questionable). Using default emission factors instead of site-specific ones led to overestimation of eutrophication impact results in the cases studied in this paper. Eutrophication impacts calculated with the most recent methods are providing lower results because more N and P attenuation/removal processes are accounted for. The farm inputs are an important determinant of the impacts, but the fate modelling (transport, attenuation) and the sensitivity of the receiving compartment is crucial. Since LCA involves inventories across many countries, the challenge is to have site-specific characterisation factors that are defined with a global coverage. That is why the models of Helmes and colleagues' for P impact on freshwater and of Cosme and colleagues' for N impact on marine water are very relevant. While regionalising both inventory results and characterisation factors is paramount for eutrophication, its lack of support in LCA commercial software is a limitation for the application of the most recent methods.

An important global issue is that the currently-accepted freshwater eutrophication indicators are only based on P. These indicators are only capturing part of the problem for water bodies that are co-limited by N and P such as Lake Taupo. Thus, there is a need to account for the contribution of N to freshwater eutrophication in such catchments. In addition, when the actual nutrient concentration is very low, the current end-point assessment method for freshwater eutrophication is not applicable. Thus, we could not use LCA as a tool to support current policies on water quality regulation for Lake Taupo. Future research needs to investigate coupling N and P fate modelling based on the most recent globally-valid and spatially-explicit LCIA methods.

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