# Application of Eutrophication Potential Indicators in a Case Study Catchment 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 Eutrophication Potential indicators in the LCA of livestock farm systems using a New Zealand (NZ) lake catchment case study. **Method:** Average dairy and sheep & beef farm systems (based on primary data) in the Lake Taupo catchment have been studied. Emissions of nitrogen (N) and phosphorus (P) to waterways from these farms have been calculated using the site-specific OVERSEER nutrient budgets model hereafter called OVERSEER. Water quality data has also been collected for the lake catchment and has defined the extent of N and/or P limitation for algal growth. In the Lake Taupo catchment, maximum catchment loads of N have been defined and used to set regulated maximum N leaching losses from farms, in order to maintain acceptable levels of water quality for the community. A range of different Eutrophication Potential indicators have been assessed and compared. These have then been related to the actual nutrient limitations of the lake to understand the effects of choice of indicator and its relevance to specific catchments. **Results:** For sheep & beef and dairy farm systems, N leaching was 13 and 49 kg N/ha/year, and P runoff losses were 1.1 and 3.0 kg P/ha/year, respectively. Ammonia emissions were also calculated. These emissions data were then used to calculate the increase in nutrients in water bodies and Eutrophication Potential using methods that vary in the inclusion of N and/or P, including ILCD (Freshwater Eutrophication Potential, Marine Eutrophication Potential calculated with ReCiPe 2008), Freshwater Eutrophication Potential calculated with ReCiPe 2015, and Eutrophication Potential calculated with CML. **Conclusions:** Freshwater Eutrophication Potential indicators that focused only on one nutrient can be inappropriate, as illustrated in NZ, where many freshwater bodies are co-limited by N and P (in terms of algal growth). Generic indicators based on P only may sometimes be irrelevant at a site-specific level. New Zealand's largest lake is a good illustration: Lake Taupo is co-limited, and water quality concerns and regulations are not focused on P, but solely on N due to increasing N levels over time. Ideally, site-specific eutrophication indicators should be used at a catchment level to account for water body specificity in LCA.

Keywords: Life Cycle Assessment, Water, Nutrients, Nitrogen, Phosphorus

## 1. Introduction

Aquatic eutrophication is one of the major water quality issues 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) (Guinee et al. 2002). An excess of these nutrients can lead to uncontrolled phytoplankton (algae) growth. In NZ, the National Policy Statement for Freshwater Management sets out the objectives and policies for freshwater management, aiming to protect quality of waterways (MfE, 2014). Minimum acceptable values for water trophic state 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 highly relevant, especially since they are the main anthropogenic source of nutrients in NZ water bodies (Scarsbrook and Melland 2015).

Owing to the significant concern about the eutrophication impacts of agriculture, Life Cycle Assessment (LCA) studies on agricultural systems often characterise this impact. Various methods exist for this impact category, but they differ in terms of inventory requirements, geographical coverage, spatial resolution, and emission pathways modelled. Eutrophication impacts calculated with CML (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 to water) are aggregated using the Redfield ratio which provides "equivalency factors" (Redfield et al. 1963). Thus, the characterisation factor is independent of whatever substances happen to be the limiting factor of algae growth in a given location (Guinee 2002). This method assumes that 100% of the emissions to water will contribute to eutrophication, meaning that the fate (transport and transformation/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 sensitive to N (i.e.: N is the limiting nutrient for marine biomass growth), whereas freshwater is considered to be sensitive to P (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 model the fate of N and P with two distinct spatially explicit models. For assessing freshwater eutrophication, the fate modelling of P was improved from a European model to a global model in ReCiPe 2015 (Helmes et al. 2012), which also assesses the persistence of P in the freshwaters. For assessing marine eutrophication, the generic fate of N is calculated for large marine ecosystems (Cosme et al. submitted).

Assuming that marine and freshwater ecosystems have distinct single limiting nutrients, being N and P, respectively, may be a methodology weakness. Indeed, Elser et al. (2007) showed that freshwater and marine ecosystems are similar in terms of N and P limitation. In NZ, freshwaters can be N-limited, P-limited or co-limited (McDowell and Larned 2008). Lake Taupo for example, NZ's largest lake, is co-limited by N and P (Pearson et al. 2016) and local government regulations are currently focused on limiting N inputs due to increasing N levels in the lake over time (WRC 2016).

The objective of this work was to evaluate different Life Cycle Impact Assessment (LCIA) methods and their relevance to estimating eutrophication impacts of freshwater using NZ's largest lake as a case study. Using case study farms, we calculated eutrophication indicators with different methods, and determined the implications of using generic and site-specific Eutrophication Potential indicators in the LCA of livestock farm systems.

## 2. Methods

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

### 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. The100 ha dairy farm has 270 cows, uses no brought-in feed and applies fertilisers at 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, with a 70:30 sheep:cattle ratio and applies fertilisers at 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 with the OVERSEER® nutrient budget model (Wheeler et al. 2007), and for ammonia and nitrous oxide, NZ-specific emissions factors from the NZ Greenhouse Gas Inventory (MfE 2015) were used (for details, refer to Table 1). OVERSEER is a nutrient model which has been validated against field site measurements from throughout NZ (McDowell et al. 2005, Wheeler et al. 2007). An attenuation factor of 50% was used for nitrate from soil (below root-zone) to freshwaters (rivers and lakes) (Elliot et al. 2014).

### 2.3. Eutrophication impact assessment

Eutrophication impacts were calculated with CML, ReCipe 2008, and ReCiPe 2015 (Huijbregts et al. 2015). We distinguished three stages in the impact assessment models; (i) the increase in nutrients in the receiving water body (the emission multiplied by a Fate Factor (FF)), (ii) divided by a reference emission in order to obtain a eutrophication potential indicator (midpoint), or (iii) multiplied by an Effect Factor (EF) as an indicator for ecosystem damage (endpoint).

*Nutrient fate modelling* – To calculate an increase in nutrients in a water body, each method relies on different inventory requirements. CML and ReCiPe 2015 rely on net emissions of nutrients to freshwater, whereas ReCiPe 2008 is based on gross supply of fertilisers and manure to agricultural soil. In other words, the FF of ReCiPe 2015 accounts for the fate of nutrients from freshwater to final receiving water compartment (freshwater or sea), whereas the $FF_{\text{gross supply}}$ of ReCiPe 2008 accounts for the fate from agricultural topsoil to final receiving water compartment (Fig. 1).

CML does not provide FF, but assumes 100% of emissions will contribute to the eutrophication potential (pathway A in Fig. 1). The use of ReCiPe 2008 FF requires caution: to account for NZ-specific volatilisation rates, we did not apply the composite FF for N to soil+air, but applied two separate FF to soil and to air (Goedkoop et al. 2009). In this study, we compared ReCiPe 2008 using impacts assessed based on gross supply (B in Fig. 1) or based on net emissions that account for our site-specific nutrient emissions modelling (C in Fig. 1). In pathway C, we applied ReCiPe 2008 by multiplying net emissions of nutrients to freshwater ($NO_3^-$) and to air ($NH_3$, $N_2O$), with $FF_{\text{net emission}}$ for nutrient emission to freshwater from a sewage treatment plant (corresponding to direct emissions in freshwater (Struijs et al. 2009)), and $NH_3$ and $N_2O$ emission to air. To apply ReCiPe 2015, we used the FF developed by Helmes et al. (2012): on average in NZ, the persistence time of P in freshwaters is 6.2 days (D in Fig. 1).

*Eutrophication potential indicator* – With CML, eutrophication potential (terrestrial and aquatic) is calculated based on an equivalency factor to convert all nutrient flows in terms of phosphate equivalents ($PO_4^{3-}{}_{eq}$) (Heijungs et al. 1992). With ReCiPe 2008, freshwater eutrophication potential is calculated using P emissions in freshwater from a sewage treatment plant as the reference emission (with an eutrophication potential equal to 1). Similarly, marine water eutrophication potential is calculated using N emitted in freshwater from a sewage treatment plant as the reference emission. Eutrophication potential assessed with ReCiPe 2015 is similar to ReCiPe 2008, but does not consider marine eutrophication anymore since an endpoint model is lacking (Huijbregts et al. 2015).

*Endpoint effects modelling* – CML method does not assess effects from a nutrient increase. ReCiPe 2008 and 2015 methods evaluate effects on freshwater eutrophication only, focusing on P, using an EF developed by Struijs et al (2011). The effect model is based on a stressor-response relationship for Dutch freshwater ecosystems. More recently, the effects modelling for P emissions has been improved by accounting for more species and freshwater types (Azevedo et al. 2013a, b). Regarding marine eutrophication (focused on N), very recent work seems promising, but is only partially published so far (Cosme et al. 2015, Cosme and Hauschild 2016) (Fig. 1).

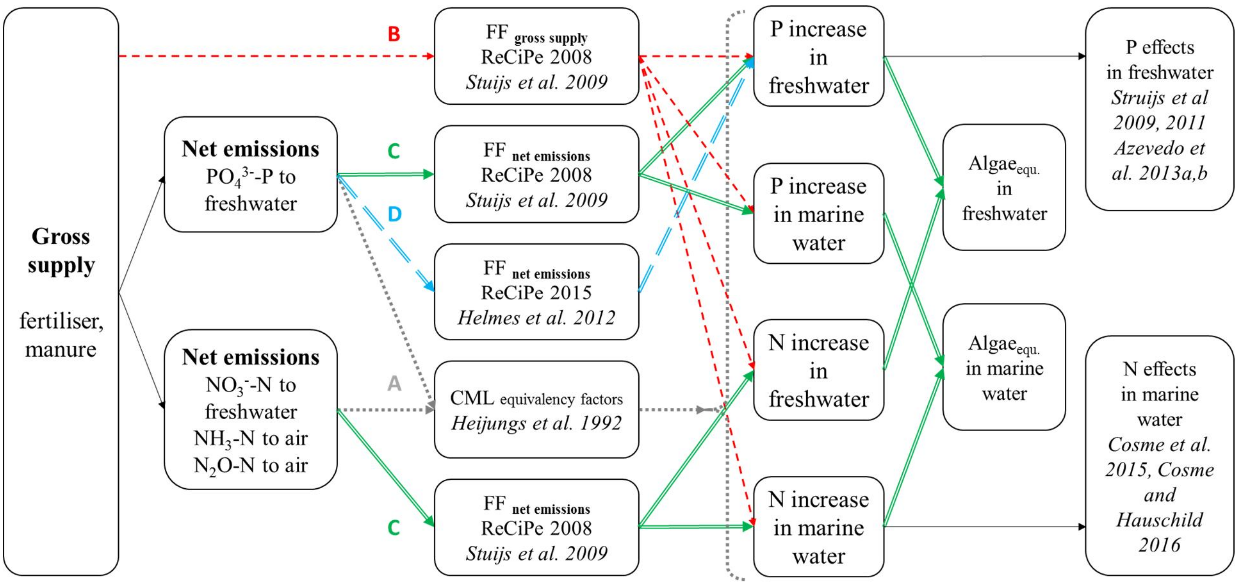


Figure 1: Nutrient inventory flow requirements, fate factors (FF) and effects modelling with different methods throughout the eutrophication cause-and-effect chain in LCA. Capital letters refer to pathways and methods applied in this work.

## 3. Results and discussion

### 3.1. Inventory of nutrient flows

Table 1 shows the inventory of nutrient flows for the sheep & beef and dairy farms. N and P emissions are 3.6 times higher per-hectare for dairy farms, but dairying land only represent less that 3% of the pastoral land area of the Lake Taupo catchment (most is in sheep & beef farming; Vant and Huser (2000). Converting sheep & beef pasture to dairying would increase the overall N load to the lake by 20-60% (Vant 2000).

Table 1: Nutrient flows inventory results [kg N or P/ha/year] and emission compartment.

| Nutrient flow | Compartment | Sheep & Beef | Dairy | Source |
|---|---|---|---|---|
| N fertiliser | agri. soil | 17 | 103 | Primary data |
| N fixation (clover/rain) | agri. soil | 63 | 122 | OVERSEER |
| N excreta dung & urine | agri. soil | 162 | 368.6 | |
| N to farm dairy effluent (FDE) | agri. soil | - | 19.4 | |
| N out in wool, meat, milk | | 19 | 62 | |
| $NO_3^-$-N leaching (below root-zone) | soil | 13 | 49 | |
| $NH_3$-N to atmosphere | air | 17.9 | 49.1 | (MfE 2015) |
| $N_2O$-N (direct) to atmosphere | air | 1.3 | 3.7 | (MfE 2015) |
| $N_2O$-N (indirect) to atmosphere | air | 0.3 | 0.8 | (MfE 2015) |
| P in excreta (manure P) | agri. soil | 15.1 | 30 | Function of DMI & P out |
| P fertiliser | agri. soil | 22 | 45 | Primary data |
| $PO_4^{3-}$-P runoff to water | freshwater | 1.1 | 3.0 | OVERSEER |
| P out in wool, meat, milk | | 3 | 11 | |

*Where DMI=dry matter intake*

### 3.2. Eutrophication impacts

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.

*Nutrient fate* – We compared fate estimates from ReCiPe 2008 using two methods, one based on gross supply of fertiliser and manure (B in Fig. 1) and the other using net emissions to air and freshwater (C in Fig. 1). N fate estimates were lower when based on net emissions (by 24% on average). The leaching fraction estimated with OVERSEER combined with the 50% attenuation factor from root zone to freshwater was less than N estimated to reach freshwaters using the CARMEN model. Also, P fate estimates were lower when based on net emissions (by 22% on average), highlighting that except for plant uptake and topsoil binding, no other P transport or attenuation process is accounted for in CARMEN, whereas our estimate of P runoff accounts for P attached to sediments lost from soil and P accumulation in the soil.

In the following section, we focus on the fate of P because ReCiPe 2008 (CARMEN) only provides an estimate of N fate, not allowing a comparison between methods. First, we compared the FF for net emissions in freshwater with the FF for gross supply of manure and fertiliser. Struijs et al. (2011) reported a difference of a factor of 18 higher for freshwater emissions of P versus agricultural emissions of P, which is similar to the factor of 20 from Huijbregts and Seppälä (2001), but is higher than the factor of seven reported by Potting et al. (2005). We cannot do a similar comparison with Helmes et al. (2012) because this method does not provide FF from agricultural emissions, since it focuses on direct emission to freshwater. We also compared the P increase in freshwater estimated with ReCiPe 2008 vs.

2015. To allow a comparison, ReCiPe 2008 FF has 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 impact using ReCiPe 2008 was 18 times higher than that using Helmes et al. (2012) (Table 2). 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 and water use modeled by Helmes et al. (2012).

*Eutrophication potential* – CML eutrophication potential corresponds to a nutrient emission expressed in phosphate equivalents, and is totally compartment-generic. Conversely, ReCiPe methods are specific to the receiving compartment, and focus on P emissions to freshwater (disregarding N emissions), and N emissions to marine water (disregarding P emissions). Thus, any comparison of these indicators would be inappropriate (Table 2).

*Endpoint effects* – It was not possible to assess effects (or damages) of an increase in nutrients in aquatic compartments with the current methods because we are outside the domain of validity of the equations for P effect (Struijs et al 2009, 2011, Azevedo et al. 2013a), and equations for N effects have just been published (Cosme and Hauschild 2016).

Table 2: Increase in nutrients in water bodies and eutrophication potential impact indicator results for different methods, expressed per ha

| **Method** | **Pathway on Fig.1** | **Impact indicator** | **Dimension** | **Compartment** | **Sheep & Beef** | **Dairy** |
|---|---|---|---|---|---|---|
| CML | A | Eutrophication potential | kg $PO_4^{3-}{}_{eq}$ | Terrestrial & aquatic | 14.38 | 42.27 |
| ReCiPe 2008 *using FF for net emissions* | | Increase in Phosphorus (Emission x $FF_{net\ emission}$) | kg P | Freshwater | 0.34 | 0.91 |
| | C | Nutrient increase and N&P aggregation in algae $_{equivalent}$ | kg algae/$km^3$ | Marine water | 0.02 | 0.06 |
| | | | kg algae/$km^3$ | Freshwater | 0.08 | 0.27 |
| | | Marine eutrophication potential (N increase/$FF_{net\ N\ emission\ to\ freshwater}$) | kg $N_{eq}$ | Marine water | 8.70 | 30.55 |
| | | Freshwater eutrophication potential (P increase/$FF_{net\ P\ emission\ to\ freshwater}$) | kg $P_{eq}$ | Freshwater | 1.10 | 3.00 |
| ReCiPe 2015 | D | Increase in Phosphorus | kg P | Freshwater | 0.019 | 0.05 |

## 4. General discussion and implications

### 4.1. Differences in inventory of nutrient flows

Default factors are not appropriate for field-specific estimates of emissions: Default volatilisation rates in ReCiPe are 21% of N in manure and 7% in fertiliser, whereas our NZ-specific volatilisation rates were 10% of N in manure and fertiliser (MfE 2015). Differences in terms of technosphere and ecosphere boundary, depending on the method, are confusing for the LCA practitioner, since they rely on different inventory requirements. There is a lack of guidelines for good practices. The second Pellston workshop on “Global guidance for LCIA indicators and methods” to be held in 2017 will help in this direction.

### 4.2. Differences in fate modelling

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 FF 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. 2016). 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 Helmes et al. (2012) method is a significant improvement toward a global nutrient fate model, but it only focuses on P and freshwater. The FF for NZ (6.2) showed a standard deviation of 19.6 days (persistence time of P in freshwater). As a result, a NZ country average FF is not appropriate: we should use a finer resolution such as the Lake Taupo catchment scale, because local hydrological properties have the largest effects on these fate factors (Helmes et al. 2012).

*N fate in freshwater* - The long time lag for leached N to groundwater of around 40 years observed in the Lake Taupo catchment (Vant 2013) is not reflected by the fate factor. This time lag is related to the deep groundwater in the catchment. In NZ, there is ongoing research on the characterisation of N attenuation according to the site-hydrogeological specificities. The reported uncertainty for attenuation factor ranges from 0 to 0.8 (Elliot et al. 2014). This uncertainty (due to natural variation of denitrification processes) has an influence on the eutrophication impact result. Future work should use site-specific fate modelling of nutrients currently under development in several catchments in NZ (Stenger et al. 2016).

*P fate in freshwater* - Sediment in lakes can act as an internal source of P, but varies with lake properties (Özkundakci et al. 2010). Similarly, Scherer and Pfister (2015) recently showed that the site-dependent P concentration in soil is one of the most important parameters influencing P emissions to water from agriculture. This illustrates the preference for use of spatially-explicit fate models.

4.3. Sensitivity of receiving water bodies

Accounting for the sensitivity of water bodies to eutrophication drivers is relevant, but doing so by focusing on a single nutrient may be inappropriate. Lake Taupo is N-and P-limited, so the freshwater effect model focused on P is only capturing part of the problem. To avoid using the concept of limiting nutrient, N and P nutrients in each receiving compartment (marine and freshwater) were aggregated using conversion factors for P and N in terms of algae, based on the Redfield Ratio (Redfield 1963 used by Goedkoop et al. 2008), assuming that one mole of algae biomass contains one mole of P and 16 moles of N (Table 2). This allows an impact indicator to be obtained that reflects an increase of both N and P nutrient in a water compartment.

4.4. Nutrient effects modelling and policy

The concentration of nutrients in Lake Taupo is lower than most European freshwaters at 0.079 $mg.L^{-1}$ total N, and 0.0052 $mg.L^{-1}$ total P on average between the years 2010 - 2014 (Vant 2013). These concentrations are so low that they are outside the domain of validity of the effect factor equation. Thus, it was not possible to assess any effect from an increase in nutrients in Lake Taupo with actual methods. The effect factor equations are valid for concentration of total P above 0.1 $mg.L^{-1}$ (Struijs et al. 2011) or above 0.05 $mg.L^{-1}$ (for temperate lake, according to Azevedo et al. 2013a). These concentrations correspond to optimum nutrient level for ecosystems, and are consistent with current water quality policies. Indeed, Struijs et al. (2011) found the highest number of species for an average total P concentration of 0.1 $mg.L^{-1}$, which is just below the regulatory water quality objectives for European lakes (0.15 $mg.L^{-1}$) (European Commission 2000). In NZ, the national bottom lines were set at 0.05 $mg.L^{-1}$ for total P and 0.75 $mg.L^{-1}$ for total N in lakes (MfE 2014). But for Lake Taupo, the objectives of water quality are more strict: the objectives is to stay below 0.0703 $mg.L^{-1}$ total N and 0.0056 $mg.L^{-1}$ 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. Nevertheless, quantifying the eutrophication impacts of dairy and sheep & beef farms in the Lake Taupo catchment is highly relevant because these farms are monitored and have a maximum nitrogen discharge allowance (WRC 2016).

## 5. Conclusions

The application of eutrophication potential indicators suffers from a lack of transparency of methods, what processes to account for, and a lack of clear guidelines of inventory requirements for LCA practitioners. The inventory of nutrient flows at a farm scale and fate factors modelled at a catchment scale should be site-specific (the relevant scale has to be determined). The farm inputs play an important role in the impact, but the fate modelling (transport, attenuation) and the sensitivity of the receiving compartment plays an important role as well. Since LCA involves inventories across many countries on a global scale, the challenge is to have site-specific characterisation factors that are defined with a global coverage.

Considering that the main currently-accepted freshwater eutrophication indicators are only based on P, we could not assess impacts on Lake Taupo, which is co-limited by N and P, and thus could not use LCA as a tool to support current policies on water quality regulation.

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