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Cosme, Nuno Miguel Dias; Niero, Monia

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Modelling the influence of changing climate in present and future marine eutrophication impacts from spring barley production

Nuno Cosme* and Monia Niero*

Division for Quantitative Sustainability Assessment, Department of Management Engineering, Technical University of Denmark, Produktionstorvet 424, 2800 Kgs. Lyngby – Denmark

* Corresponding authors.
E-mail addresses: nmdc@dtu.dk (N. Cosme), monni@dtu.dk (M. Niero)

Abstract

Nitrate concentration and runoff are site-specific and driven by climatic factors and crop management. As such, nitrate emissions may increase in the future due to climate change, affecting the marine eutrophication mechanism. In this context, and considering the case of spring barley production in Denmark, the paper has two objectives: (i) to estimate the present and future marine eutrophication impacts by combining a novel Life Cycle Impact Assessment (LCIA) modelling approach with a quantification of the effects of climate change on its parameterisation, and (ii) to discuss the implications of different normalisation references when comparing future Life Cycle Assessment (LCA) scenarios with current production systems. A parameterised characterisation model was developed to gauge the influence of future climatic-driven pressures on the marine eutrophication impact pathway. Spatial differentiation was added to the resulting 'present' and 'future' characterisation factors (CFs) and calculated for the Baltic and North Sea. The temporal variability of both midpoint normalised impact scores and damage scores reflect a 34% and 28% increase of the CFs in the North Sea and Baltic Sea, respectively. The temporal variability is mostly explained by CF variation and increasing future nitrogen flows. The marine eutrophication indicator scores at both midpoint and damage levels suggest that the differentiation of impacts to various receiving (and potentially perturbed) ecosystems is relevant. Damage scores are quantified with a factor 2.5 and 2.3 differentiation between the Baltic (higher) and North Seas (lower) for the present and future scenarios, respectively. The comparison of the normalisation methods, either based on total annual impacts (domestic inventory of background interventions), on ecological carrying capacity, or on the presently proposed method, point to the value of adding spatial differentiation to LCIA models. The inclusion of time variation and spatial differentiation in characterisation modelling of marine eutrophication and the identification of a paucity of adequate inventory data for future scenario analysis constitute the main outcomes of this study. Further research should aim at reducing the uncertainty of the parameterisation under future conditions and strengthening emissions projections.

Keywords

Life Cycle Assessment; LCIA; Damage modelling; Normalisation; Marine ecosystems; Climate change.

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1. Introduction

Conserving and sustainably using the oceans, seas and marine resources, taking urgent action to mitigate and adapt to climate change, and achieving food security whilst improving nutrition are three of the 17 Global Goals 2030 Agenda for Sustainable Development Global Goals (UN General Assembly, 2015). Global food security and environmental sustainability are interlinked, whereby the former only becomes possible if agricultural systems meet certain sustainability criteria (Foley et al., 2011). Life Cycle Assessment (LCA) is one method to holistically assess whether agricultural systems are meeting the necessary benchmarks. The use of LCA to assess the potential environmental impacts of agricultural systems is growing (Soussana, 2014), and guidance on tailoring LCAs for crops has recently been published, with regard to the agri-food sector (Notarnicola et al., 2015). Agriculture and energy production are the main sources of environmental emissions of reactive nitrogen (N) (Galloway et al., 2008). The application of fertilizers in agriculture introduces ammonium (NH_4^+) and nitrate (NO_3^-) to soil and water, and ammonia (NH_3) to air, whereas the combustion of fossil fuels adds nitrogen oxides (NO_x) to air (Socolow, 1999). In agriculture practices, N added to the soil may exceed plant assimilation. This surplus emitted to the

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environment may constitute the main cause for anthropogenic fertilization of freshwater and marine ecosystems that lead to deleterious aquatic eutrophication.

Marine eutrophication is a syndrome of ecosystem responses to the increase of the availability of growth-limiting plant nutrients in the euphotic zone of marine waters (Cloern, 2001; Cloern et al., 2016; Nixon, 1995; Smith et al., 1999). For modelling purposes, nitrogen is assumed to be the growth-limiting nutrient in marine waters, considering representative average spatial and temporal conditions (see also Vitousek et al. (2002); Howarth and Marino (2006); Cosme et al. (2015)). Such N-enrichment promotes planktonic growth and often involves depletion of dissolved oxygen (DO) in bottom waters to hypoxic and anoxic levels, potentially affecting exposed species (e.g. Gray et al., (2002); Levin et al., (2009), Vaquer-Sunyer and Duarte, (2008)). Impacts of eutrophication-induced hypoxia are seen from the local to regional scales (Breitburg et al., 2009). Similarly, variability at short time scales (e.g. seasonal) can have a significant role in impacts modelling, e.g. latitude and light availability, temperature and species distribution, water stratification and oxygen depletion. Current research of Life Cycle Impact Assessment (LCIA) methods for marine eutrophication is being directed to improve the representation of short term variability and spatial differentiation – see e.g. Azevedo et al. (2013); Cosme and Hauschild (2016a, 2016b); Cosme et al. (2016a, 2016b, 2015). Parameterisation of future pressures in those methods for impact forecasting is naturally absent. To the knowledge of the authors no other studies addressing the effects of time variation and future environmental conditions on marine eutrophication in LCA exist.

Nitrate concentration and runoff are site-specific and driven by climatic factors and crop management, as shown for organic cereal cropping systems in Denmark (Jabloun et al., 2015). As a consequence of the expected increase in temperature and changed rainfall pattern, N runoff may increase in the future (Doltra et al., 2012; Jensen and Veihe, 2009). However, to what extent N and water management can close the yield gaps is still uncertain (Mueller et al., 2012). Therefore the definition of future scenarios for agricultural systems is not straightforward.

In the LCA framework, future-oriented scenarios for crop production have so far mainly focused on comparing different GHG mitigation options of both crops and livestock production on farms in northern Europe and USA (Audsley and Wilkinson, 2014), wheat in the UK (Röder et al., 2014), as well as to compare different adaptation strategies, e.g. for wheat in Switzerland (Tendall and Gaillard, 2015) and UK (El Chami and Daccache, 2015), and for barley in Denmark (Dijkman et al., 2013; Niero et al., 2015b). Guidance to manage uncertainty in the definition of future LCA scenarios addressing the effect of climate change in crop production is provided at the Life Cycle Inventory (LCI) level and implemented in the case of spring barley cultivation in Denmark under a future, realistic, worst-case climate scenario (Niero et al., 2015a). However, the effect of increased temperature and CO_2 concentration will also affect the impact pathway and therefore the LCIA modelling. A similar approach using temporal scenarios (present and future) to address the influence of climate change at the regional scale has been applied for water availability (Núñez et al., 2015).

Marine eutrophication characterisation in LCIA models the variation of an indicator located between the emission and the damage through an impact pathway, e.g. dissolved N concentration increase, as in the ReCiPe (Goedkoop et al., 2012), EDIP 2003 (Hauschild and Potting, 2005), IMPACT 2002+ (Jolliet et al., 2003), and CML 2002 (Guinée et al., 2002) LCIA methods. Marine eutrophication indicators at a later point (closer to the damage) would need a longer modelling work of the environmental mechanisms, but are lacking in the methods above. The inclusion of ecosystem exposure and effects on biota, as done for the ecotoxicity indicator (Rosenbaum et al., 2008), is proposed here for marine eutrophication – see also Cosme and Hauschild (2016b). The impact assessment, at any point, is done by applying substance-specific characterisation factors (CF) that convert the emissions into a potential impact (Hauschild, 2005).

The characterisation modelling work of the marine eutrophication indicator presented here was developed in the EU FP7 project LC-IMPACT (http://lc-impact.eu/) and was improved with recent developments. It involves the estimation of CFs consistent with the generic impact assessment framework (Udo de Haes et al., 2002) by modelling factors for the environmental fate of emissions, ecosystem exposure to these, and effects on exposed species.

Normalisation in LCA relates the characterised impact indicator scores of an analysed system to those of a reference system (Laurent and Hauschild, 2015). It is an optional step in the characterisation phase and it is useful to understand the relative magnitude of the impact indicator (ISO 14044, 2006). Different normalization references can be applied, with different reference duration of the included activities and boundaries of the reference system, i.e. following either a production-based or a consumption-based perspective. In both cases, the flows from all activities occurring within the physical or geographical boundaries of the reference system over the reference duration need to be quantified, either in terms of the total production activities or total consumption of the reference system, respectively (Laurent and Hauschild, 2015).

Building on the results of an LCA study of spring barley in Denmark (Niero et al., 2015b), this paper estimates the present and future marine eutrophication impacts by combining a novel LCIA approach which includes the influence of climate change using model parameterisation to add both temporal and spatial variation beyond previous attempts. Furthermore, the implications of different normalisation references when comparing future LCA scenarios with current production systems are discussed.

2. Materials and methods

First, the framework to characterise the marine eutrophication impact category is introduced (section 2.1) and the LCI data used to feed the LCIA model are presented (section 2.2). Secondly, the parameterisation in the LCIA under present and future climate conditions is presented, including the implications of climate change on marine eutrophication modelling (section 2.3), as well as the possible adaptations of normalisation procedures in future scenarios definition (section 2.4), and a method to estimate damage factors for marine eutrophication damage modelling (section 2.5).

2.1. DPSIR and impact pathways for marine eutrophication

Environmental indicators have become an important tool in decision-making (Tscherning et al., 2012), often benefiting from conceptual frameworks based on causality (Niemeijer and Groot, 2008). The causal chain framework Drivers-Pressures-State-Impacts-Responses (DPSIR) (Smeets and Weterings, 1999) is formally an adaptive environmental management approach that integrates environmental and human systems into a common conceptual framework.

The Drivers can be defined as economic and social factors triggering Pressures to the environment (Borja et al., 2006). Applying the DPSIR approach to the marine eutrophication impacts indicator (Figure 1), the primary Drivers arise from the population growth and consequent need for food and energy (Galloway et al., 2008; Zaldívar et al., 2008). The Pressures express the way ecosystems are disturbed by human activities (Borja et al., 2006), and correspond to the N emissions identified in the LCI. The State refers to the ecosystem condition under the Pressures, and can be assessed by field measurements or indicators (Bricker et al., 2008; Ferreira et al., 2011). Impacts are the effects on the ecosystem and society caused by changes in the State, like hypoxia that causes behavioural, physiological, or ecological impacts on biota (e.g. Davis (1975), Diaz and Rosenberg (1995), Gray et al. (2002), Vaquer-Sunyer and Duarte (2008)), or like toxic and harmful algal species, loss of biodiversity, water quality degradation hindering water uses, fish production, or aesthetic value (Kelly, 2008; Rabalais, 2002). The Responses are the management and societal measures aimed at preventing, minimising, or mitigating the Impacts by feeding back to the D-P-S, i.e. modifying the Drivers, reducing Pressures, and restoring the State to 'healthy' conditions.

LCIA indicators focus on the P-S-I components, based respectively on inventoried emissions in LCI, fate (on P) and exposure (on S) modelling work, and the effects modelling (on I). The LCA framework supports decision-making processes in devising Responses aimed at modifying the Drivers and reducing the Pressures. The conceptual 'management sphere' thus feeds back information and action from the problems in the ecosphere to the solutions in the technosphere, which is the core value of LCA – the characterisation of the interface between techno- and ecosphere.

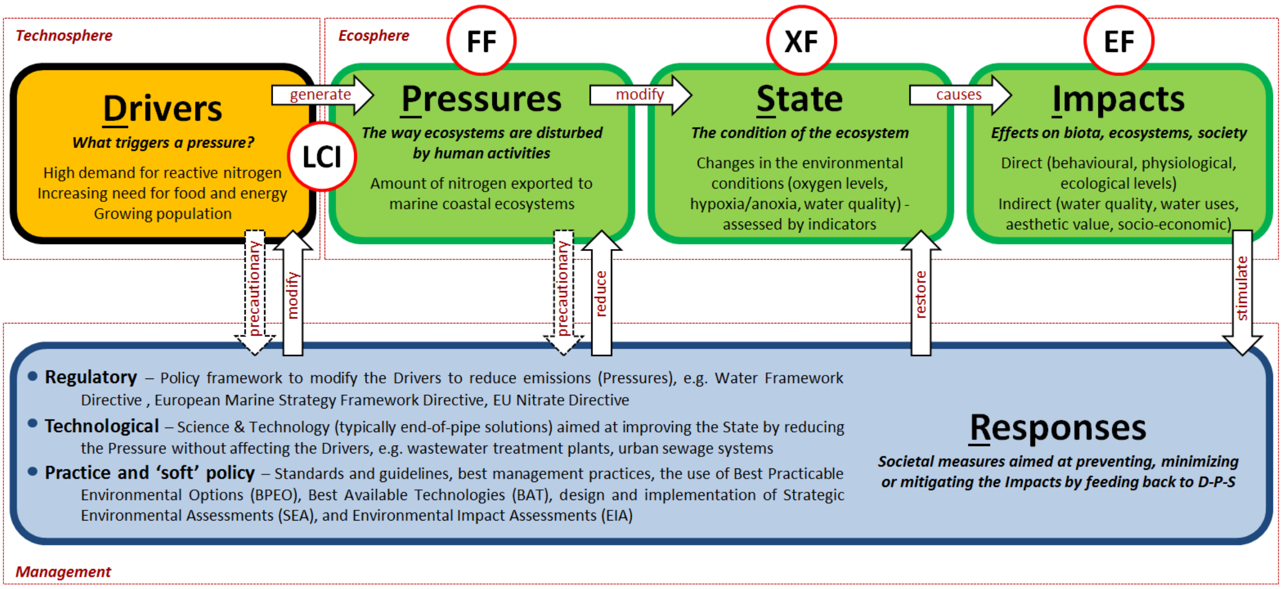

Figure 1 The Drivers-Pressures-State-Impacts-Responses (DPSIR) framework applied to the marine eutrophication indicator in life cycle impact assessment. Indication of the impact assessment modelling components and interface with the DPSIR framework: fate factor (FF), exposure factor (XF) and effect factor (EF) are combined in order to characterise emissions inventoried in the life cycle inventory (LCI) phase.

2.2. Life Cycle Inventory of present and future spring barley cultivation

The details of the scenario describing the present spring barley cultivation in Denmark are reported in Niero et al. (2015b). This scenario, with 'cradle-to-farm gate' boundaries, refers to the average cultivation of 1 kg of dry matter spring barley (*Hordeum vulgare* L.) grain for malting in Denmark (functional unit). The average Danish crop yield in the 5-year interval 2009–2013 was considered (5,700 kg·ha^{-1}). For future spring barley cultivation, the data on crop yields produced in the climate phytotron RERAF (Risø Environmental Risk Assessment Facility) were used, where spring barley cultivars were cultivated under controlled and manipulated treatments mimicking a worst case climate change, i.e. double CO_2 concentration (700 ppm) and a global mean temperature increase of 5°C in the atmosphere (Ingvordsen et al., 2015). The measured variation in crop yield depends on the set of cultivars and experimental conditions (Niero et al., 2015b), but it is considered here equal to 4,207 kg·ha^{-1} (26% less than current situation). In the experiments mimicking future climate the amount of fertilizer currently applied was used, therefore the amount of N·ha^{-1} was kept constant for the future scenario, but assuming an increase in nitrate leaching (+24%) (Jensen and Veihe, 2009). The LCI model delivers emissions of NO_3^- to water and NH_3 and NO_x to air calculated per ha of cultivated land and kg yield (Table 1). The calculation of the N emissions described above are based on emission factors model work by Hamelin et al. (2012) (for NO_x and NH_3) and Kristensen et al. (2008) (for NO_3^-) and the N content in fertilizer – see details in Niero et al. (2015b).

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Table 1 Summary of emitted quantities of N-derived substances per emission route, in the present and future spring barley production system (based on Niero et al. (2015b)).

Elementary flow	Amount emitted		Unit
	Present scenario	Future scenario	
N in nitrogen oxides (NO_x-N) to air:			
- per area	1.77	1.77	$kg \cdot ha^{-1}$
- per yield	9.88E-05	1.34E-04	$kgN \cdot kg_{barley}^{-1}$
N in ammonia (NH_3-N) to air:			
- per area	7.34	7.34	$kg \cdot ha^{-1}$
- per yield	1.06E-03	1.43E-03	$kgN \cdot kg_{barley}^{-1}$
N in nitrate (NO_3^--N) to water:			
- per area	126	157	$kg \cdot ha^{-1}$
- per yield	4.99E-03	8.43E-04	$kgN \cdot kg_{barley}^{-1}$

2.3. Characterisation factors for marine eutrophication under present and future scenarios

The impact assessment methodology characterises waterborne N emissions as nitrate (NO_3^--N) and airborne N deposition as ammonia (NH_3-N) and nitrogen oxides (NO_x-N) obtained in the LCI (Table 1). The characterisation model used here applied LC-IMPACT marine eutrophication CFs modified with recently developed XF and EF models. The CF is composed of a Fate Factor (FF) that quantifies the environmental losses from the original emission in freshwater and marine compartments expressing the availability of N in the euphotic zone of coastal waters (Azevedo et al., 2013), an eXposure Factor (XF) that expresses the 'conversion' potential of the available N into organic matter (biomass) and oxygen consumed after its aerobic respiration (Cosme et al., 2015), and an Effect Factor (EF) that quantifies the effect of oxygen depletion on exposed species (modelled as time- and volume-integrated Potentially Affected Fraction of species, PAF) (Cosme and Hauschild, 2016a).

The emitted amounts of N from each of the emission routes (e.g. to air, surface freshwater, groundwater, or marine water) are multiplied by the respective CF to deliver the impact score (IS) for the specific human activity from which the reported emission was originated, per receiving marine coastal ecosystem (66 spatial units). The Large Marine Ecosystems (LME) biogeographical classification system (Sherman and Hempel, 2009) was adopted for its consistent use in the three factors modelled. Coastal ecosystems LME#22 (North Sea) and LME#23 (Baltic Sea) were identified as the receiving coastal spatial units for Danish emissions.

Predictions of future pressures caused by altered climatic conditions predominantly describe negative consequences for biodiversity and ecosystems functions (Brierley and Kingsford, 2009; Rabalais et al., 2009). Modelling such future impacts involves a highly uncertain quantification of both pressures and responses (biogeochemical, biological, and ecological) due to the diversity of potential impacts and the complexity of cumulative and synergistic effects. For this reason, caution should be applied to its application and especially interpretation.

The major drivers for those pressures relate to increased temperature, sea level rise, enhanced hydrological cycles, and shifts in wind and currents patterns (Rabalais et al., 2009). Individually, or cumulatively, these impose direct and indirect effects on species and ecosystems. Increased temperature directly affect physiological aspects such as increasing metabolic rates, including oxygen requirements, temperature or hypoxia stress, heterotrophic respiration and oxygen consumption (Pörtner and Knust, 2007; Rabalais et al., 2009); or indirectly, via phenology and species succession by altering food availability and food webs (Edwards and Richardson, 2004). On the abiotic component, temperature- and salinity-driven density gradients (pycnoclines) may be strengthen, with special impact on intensified stratification in coastal waters (Rabalais et al., 2009). Stratification hinders oxygen diffusion and vertical mixing, facilitating the onset of hypoxia in bottom waters and the disruption of biogeochemical cycles (Diaz and Rosenberg, 2008; Middelburg and Levin, 2009). Moreover, oxygen solubility in seawater is a function of temperature. In a future warmer ocean altered availability of oxygen may pose important limitations to species occurrence (Brierley and Kingsford, 2009). In addition, increased riverine discharge of nutrients and organic matter, from a potential increased precipitation regime, may exacerbate oxygen depletion after its respiration in shallow coastal waters. Reviews of these and other future pressures and effects can be found in (Brierley and Kingsford, 2009; Rabalais et al., 2009).

In an attempt to model the influence of the pressures affected by future climate change and to add environmental relevance to the characterisation modelling of the future scenario, modifications to the parameterisation of the original CFs were introduced (Table 2).

Table 2 Changes introduced in the characterisation modelling factors in order to represent the influence of future climatic-driven pressures. Abbreviations used: fate factor (FF), exposure factor (XF), effect factor (EF), North Sea (NS), Baltic Sea (BS), climate zone (CZ).

Parameter	Induced change	Driver for change	Affected factor	Reference
Mean annual sea surface temperature	From 10.5°C to 12.3 °C (NS), from 8.3°C to 9.8°C (BS)	Temperature increase	FF, XF	Belkin (2009); Cosme et al. (2015)
Mean annual bottom water temperature [a]	From 10.5°C to 12.3 °C (NS), from 8.3°C to 9.8°C (BS)	Temperature increase	FF, XF	Cosme and Hauschild (2016a)
Q_{10}, Temperature Coefficient (increase factor of a rate at a 10° temperature increase)	$Q_{10} = 2$	Temperature increase	FF, XF	Söderlund and Svensson (2012)
Nitrogen removal rate in freshwater systems	From 0.527 to 0.595 (NS) and 0.584 (BS) removal fractions (Q_{10}-based)	Temperature increase	FF	Wollheim et al. (2008)
Residence time in coastal waters	Constant	Altered wind and hydrographic patterns	FF	-
Denitrification rate in marine compartment	From 0.3 to 0.338 (NS) and to 0.332 (BS) denitrified	Temperature increase	FF	Van Drecht et al. (2003)

Parameter	Induced change	Driver for change	Affected factor	Reference
	fractions (Q_{10}-based)			
N losses by advection in marine compartment	Constant	Altered wind and hydrographic patterns	FF	-
Respiration rate of sinking marine snow [b]	From 0.13 d^{-1} to 0.145 d^{-1} (Q_{10}-based)	Temperature increase	XF	Iversen and Ploug (2010)
Phytoplankton grazed fraction (f_{PPgrz})	10% shift from sink to grazed fraction: f_{PPgrz} from 0.3 to 0.27 (NS), and 0.49 to 0.44 (BS)	Temperature increase	XF	Cosme et al. (2015)
Bacterial Growth Efficiency (metabolic rate)	From 0.22 to 0.248 (NS), and 0.37 to 0.421 (BS) (Q_{10}-based)	Temperature increase	XF	Cosme et al. (2015)
Species poleward shift	20% influence of species from temperate CZ (NS) and 10% (BS) on sensitivity to hypoxia	Temperature increase, wind and currents patterns, advection	EF	Cosme and Hauschild (2016a)

[a] Continental shelf depth is assumed as of 200 m; for modelling purposes the average depth is 100 m (Cosme and Hauschild, 2016a).
[b] Marine snow refers to the sinking flux of particulate organic carbon (POC) of aggregates of phytoplankton cells, faecal pellets, zooplankton carcasses, and other organic material from dead or dying microorganisms (Fowler and Knauer, 1986).

2.4. Normalisation under present and future scenarios

The years chosen to be representative of the current and future scenarios are 2010 and 2050, respectively. Characterised impact scores at the midpoint level (mpIS) were normalised with an external normalisation reference (NR) (production-based, per capita). This was calculated with the same LC-IMPACT marine eutrophication characterisation model applied to the annual emissions from inorganic fertilisers and manure in 2010 in Denmark using a nitrogen use efficiency coefficient of 0.4 and N-content in annual applications (Bouwman et al., 2009), sewage water in 2010 following the emission model by Van Drecht et al. (2009), and NO_x-N and NH_3-N in 2005 after Roy et al. (2012). The NR for the future scenario (year 2050) was estimated from projections of fertilizers application (FAOSTAT, 2013), GDP growth in Denmark (TradingEconomics, 2015), and predicted future emissions of NO_x and NH_3 in Denmark (Nielsen et al., 2014). The calculated NRs for 2010 and 2050 are included in Table 3.

Marine eutrophication emerged as one of the most contributing impact categories for the current spring barley cultivation scenario after normalisation performed with the ReCiPe LCIA method at midpoint level (Niero et al., 2015a). It is also one of the impact categories showing the highest variation from current to future scenario (Niero et al., 2015a). It would be interesting to verify whether the situation is confirmed also under future climatic pressure, but currently there are no available characterisation models and normalisation references that cover future pressures for marine eutrophication. Therefore, different approaches to normalisation at the midpoint level were compared, referring to the recommended ILCD LCIA methodology (Hauschild et al., 2013). One, was the traditional normalisation approach, where the indicator scores of a product system are compared to those of society's background interventions, i.e. the EU-27 'domestic inventory' in 2010 corresponding to the emissions and consumptions in that spatial and temporal scope (Sala et al., 2015). An alternative normalisation reference was also included, based on the carrying capacity of ecosystems, i.e. the maximum environmental intervention these can withstand without experiencing negative changes, recently proposed by Bjørn and Hauschild (2015). In such approach, NRs were calculated as the carrying capacity for each impact category divided by the population in the reference region and year. For the future scenario, the population in Europe (EU-28) in 2050 was used. A summary of the considered scenarios and data/assumptions in the calculations is reported in Table 3.

Table 3 Summary of the inventory data used to calculate the normalisation references (NR) for Denmark (DK) in 2010 and 2050.

Reference	NR LC-IMPACT		NR 'domestic inventory' [1]		NR carrying capacity [2]	
Scope	DK (2010)	DK (2050)	EU-27 (2010)	EU-28 (2050)	EU-27 (2010)	EU-28 (2050)
Background intervention (kgN·yr^{-1})	3.55E+09	3.98E+09	8.44E+09	1.12E+10	-	-
Carrying capacity (kgN·yr^{-1})	-	-	-	-	2.27E+10	2.27E+10
Population (pers) [3]	5,417,692	6,271,485	498,867,771	525,527,890	498,867,771	525,527,890
NR value (kgN·pers^{-1}·yr^{-1})	641	620	16.9	21.3	45.6	43.3

[1] Source: Sala et al. (2015);
[2] Source: Bjørn and Hauschild (2015);
[3] Source: DK 2010 and EU-27 2010 – EUROSTAT (2015a); DK 2050 and EU-28 2050 – EUROSTAT (2015b).

2.5. Damage factors

Midpoint modelling was extrapolated to damage level by converting PAF to Potentially Disappearing Fraction (PDF) of species and by applying spatially explicit species densities. The metrics conversion and the species density-based weighting corresponds to the damage factor (DF). This approach is also adopted in the ReCiPe method (Goedkoop et al., 2012), but the spatial differentiation feature is limited to a single site-generic marine species density value.

For the PAF to PDF metrics conversion a factor 0.5 was chosen, i.e. PDF=0.5*PAF, as discussed in Cosme et al. (2016a) (see also Jolliet et al. (2003) and Larsen and Hauschild (2007)), or the assumption that 50% of the species affected eventually disappear due to hypoxic stress. Species density (SD, in species·m^{-3}) then converts PDF into species·yr – the unit for 'Ecosystems' damage in the ReCiPe method (Goedkoop et al., 2012). Spatially explicit species density values are available per LME as 6.7E-12 species·m^{-3} in the North Sea and 3.6E-12 species·m^{-3} in the Baltic Sea (Cosme et al., 2016a).

3. Results and discussion

Present and future inventory flows from spring barley production were characterised with the proposed spatially explicit CFs for the marine eutrophication indicator. Results were normalised with three alternative methods and analysed. Indicators of damage to ecosystems were further calculated for the same temporal scenarios. The results of these estimations are presented and discussed in the next sections.

3.1. Characterisation factors under present and future scenarios

The CFs applied to the present and future spring barley scenarios in the various routes and receiving LMEs are included in Table 4. For the present scenario, N emissions from spring barley cultivation (Table 1, second column) were characterised using the spatially differentiated FF, XF, and EF (see section 2.3). For the future scenario, future N emissions (Table 1, third column) were characterised using the modified FF, XF, and EF parameterised in accordance to the influence of future climatic-driven pressures, as reported in Table 2.

Table 4 Marine eutrophication characterisation factors (CFs) used to characterise present and future nitrogen (N) emissions from spring barley production to the North Sea and Baltic Sea, estimated from fate factors (FF), exposure factors (XF), and effect factors (EF) modelling.

Scenario		Present		Future	
Receiving ecosystem		North Sea	Baltic Sea	North Sea	Baltic Sea
Factor	Emission route				
FF [yr]	NO_3^--N to water	0.59	1.39	0.48	1.12
	NO_x-N to air	0.05	0.12	0.04	0.10
	NH_3-N to air	0.05	0.12	0.04	0.10
XF [$kgO_2 \cdot kgN^{-1}$]	All	9.11	15.9	8.30	13.91
EF [$(PAF) \cdot m^3 \cdot kgN^{-1}$]	All	1.59	1.78	1.70	1.91
CF [$(PAF) \cdot m^3 \cdot yr \cdot kgN^{-1}$]	NO_3^--N to water	8.53	39.20	6.81	29.76
	NO_x-N to air	0.75	3.46	0.60	2.62
	NH_3-N to air	0.74	3.39	0.59	2.57

The future FFs are lower than present FFs due to a predicted increase of the denitrification rate in both freshwater and marine compartments (Veraart et al., 2011). This fact leads to a lower N-fraction available to promote eutrophication impacts (Cosme et al., 2015). The XFs decrease in the future scenarios due to i) a predicted larger fraction of phytoplankton grazed and less sinking material to be respired near the bottom, and (ii) increased metabolic rates (with enhanced respiration of sinking marine snow dominating the enhanced bottom respiration). In both cases, oxygen depletion and eutrophication potential are decreased (Cosme et al., 2015). The future EFs predict higher impacts as species shift poleward from the Celtic-Biscay shelf (ca. 14% and 23% more sensitive to hypoxia than North Sea's and Baltic Sea's, respectively) (Cosme and Hauschild, 2016a). Given the higher variation of the XF to the CF estimation (Table 4) and the potential underestimation of future pressures in the EF modelling, these are believed to be the most relevant sources of variation in the future CFs. Other possible future pressures, not quantified in Table 2 due to high uncertainty, may change habitat conditions and lead to significant increase of the CFs, like stronger water stratification and reduced oxygen solubility that affect, respectively, the XF and EF.

Acknowledging the concerns about uncertainty in modelling both present and future CFs, the advantage of producing spatially explicit impact scores to LCIA seems highly relevant (Potting and Hauschild, 2006; Udo de Haes et al., 2002). Moreover, given the spatial differentiation of species distributions at the same scale (i.e. LME-dependent), these can be coupled for the damage modelling.

3.2. Uncertainty in the normalisation step

Figure 2A shows normalised impacts scores (normIS) for marine eutrophication at the midpoint level (characterised with ILCD recommended CFs, i.e. ReCiPe's CFs for aquatic eutrophication applied to N flows) using 'domestic inventory' NRs and carrying capacity NRs for Europe, and LC-IMPACT NRs for Denmark, in 2010 (present scenario) and 2050 (future scenario).

The mpIS obtained with the ILCD LCIA method for present and future emissions from the spring barley production system are based on the same characterisation model, i.e. use the same CFs (Niero et al., 2015b). It is assumed that such model is adapted to represent the present impacts. The model fit for future conditions is not quantified or discussed here, because the underlying models have no parameterisation adjustment to represent the effect of future pressures. The normalisation references calculated for the future scenario (3) adopt reference emissions inventory (background interventions) and population values for 2050, but use the same characterisation model as for 2010, introducing an inevitable uncertainty to the normIS of the future scenario.

The adoption of the alternative normalisation based on carrying capacity helps quantifying that misestimation. Beyond short-timed natural variability, the carrying capacity is per definition constant at the timescale used here (decades) (Bjørn and Hauschild, 2015), so it can be assumed that there is no additional uncertainty introduced in the normIS. The 'domestic inventory'-based and carrying capacity-based NRs vary in their essence, i.e. relative to a varying (yearly) background in the former and to a fixed (European) carrying capacity in the latter (the contribution from the population increase is the same in both methods). The variation in magnitude (Figure 2A) is justified by the carrying capacity being 2.6 times higher than the 'domestic intervention' in 2010 and 2 times in 2050 (3), whereas the variation in relative contribution (Figure 2B) originates from considering a population growth in the 'domestic inventory' NR_{2050} but a constant carrying capacity value in this method (3).

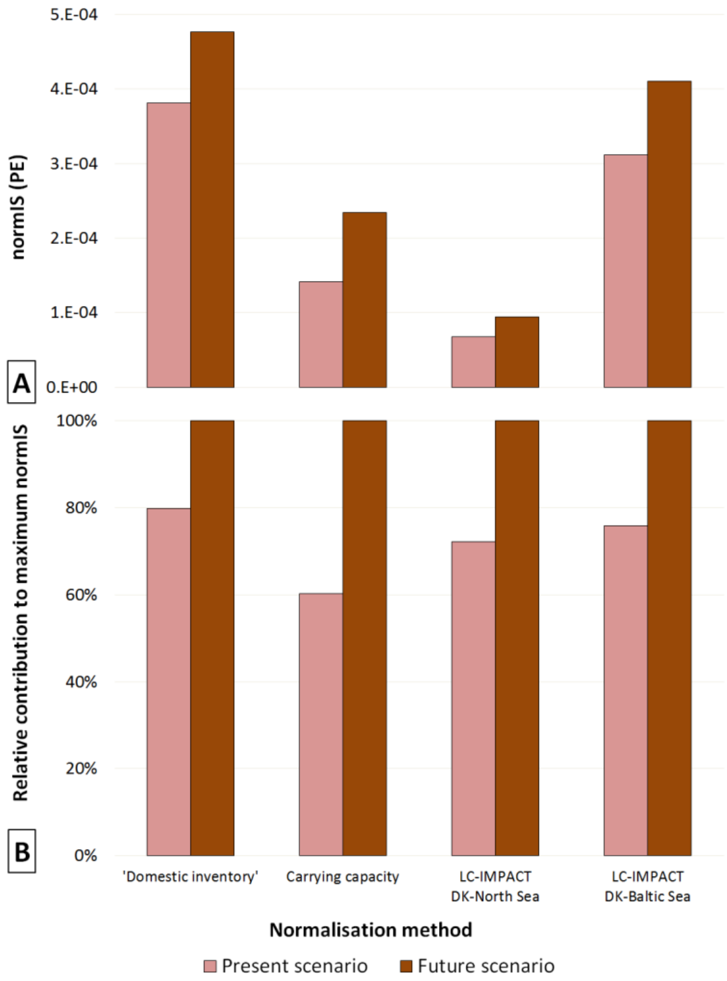

Figure 2 A) Normalised impacts scores (normIS) for marine eutrophication at midpoint level, B) Relative contribution (in %) of each normIS to the maximum score. Both sets of results calculated for the present scenario (2010) and future scenario (2050) per normalisation method used – the EU 'domestic inventory' and carrying capacity-based NRs for Europe, and LC-IMPACT NR for Denmark.

Since the midpoint LC-IMPACT-based CFs model a longer marine eutrophication impact pathway, those mpIS are therefore not comparable to ILCD's (units are (PAF)·m^3·yr and kgN-eq, respectively). The normalisation step eliminates any uncertainty in the characterisation modelling and the results can be compared. The normIS show an increase in both ecosystems, i.e. 0.04 to 0.06 PAF·m^3·yr (North Sea), and 0.20 to 0.25 PAF·m^3·yr (Baltic Sea). The normalisation step also reveals the steeper increase in the Baltic Sea (Figure 2A) due to the spatial differentiation feature embedded in the model. This is particularly visible in the FF (4.6 times higher for the Baltic Sea and North Sea, Table 4) or 2.6 times the variability of the XF and 4.1 the EF's. The present and future LC-IMPACT NRs are very similar (section 2.4), due to the cancelling effect of increasing waterborne N emissions (inorganic fertilizer, manure, and sewage discharge) but decreasing airborne emissions (NO_x and NH_3). As such, the variability of the normIS results in the future scenario is mostly explained by (i) CFs variation (+34% and +28% from present to future CFs, for the North Sea and Baltic Sea, respectively, Table 4), (ii) the larger LCI flows, and (iii) the population change (+16%)

projected for 2050 in Denmark. The contribution of these three terms to the total uncertainty of the characterisation model is not quantified here. However, the high sensitivity to the XF and EF, and the confidence on the projections for 2050 (especially in the quantification of the total annual emissions) are potentially determinant in explaining the variability of the characterisation model and normalisation step, respectively. Despite the overall uncertainty of the marine eutrophication model in the modified LC-IMPACT method, it seems valuable to i) add environmental relevance, by including the effect of future climate pressures in the characterisation model expressed in the future impact scores, and ii) increase the completeness of the impact pathway coverage in modelling a later midpoint indicator that includes the ecosystem exposure and the effect components in the model.

The total N emissions in Denmark in 2050 were split evenly towards the North Sea and the Baltic Sea for the characterisation step – this procedure is a necessary simplification in the method at this point but may add a significant uncertainty in the normalised scores. The variation of normIS from present to future emissions (Figure 2A) shows relative increases similar to ILCD-based method (the currently recommended method that used ReCiPe's aquatic eutrophication midpoint model for N emissions), therefore suggesting that future N emissions from Denmark follow those of the European average in 2050.

The results presented in this assessment do not intend to give a full perspective of the environmental profile of the spring barley production as other impacts indicators are lacking. Similarly, the discussion is not on the sustainability of the spring barley production system (see Niero et al., 2015b), but rather on the value of introducing temporal and spatial variation in the impact assessment model.

The LC-IMPACT NRs show the relevance of introducing spatial differentiation, especially for indicators of local to regional impacts. These NRs are estimated from present and future emissions normalised by the respective present and future national emissions per capita. In opposition, the 'domestic inventory' based NR for the future scenario, are inconsistently representing the reference system, as no projection of this inventory is available so far. The carrying capacity-based NRs use a constant global carrying capacity, so the NRs variation is directly dependent on the reference system's emissions.

3.3. Damage modelling scores and application

The results of the damage scores estimation (Figure 3), based on midpoint characterisation of barley production emissions (Niero et al., 2015b) and DF application (section 2.5), show an increase of damage towards future conditions in both receiving marine ecosystems considered. Damage indicators show a factor 2.5 and 2.3 of spatial differentiation between the Baltic (higher) and North Seas (lower) for the present and future scenarios, respectively. Such differentiation is mostly caused by the higher primary productivity potential of the Baltic Sea (Cosme et al., 2015).

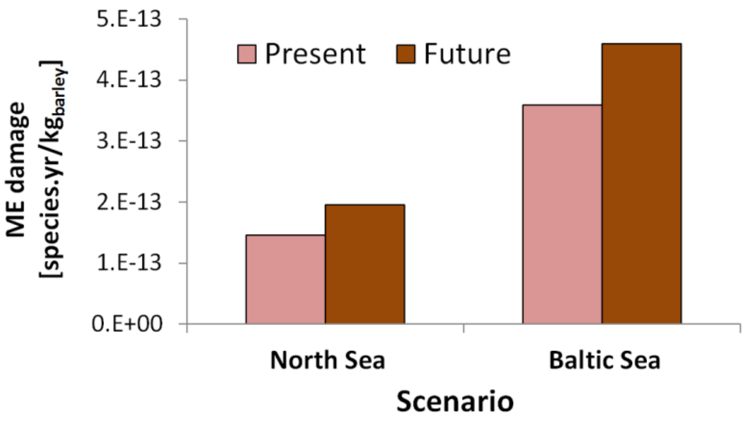

Figure 3 Damage impact scores to marine eutrophication (ME) for the present (2010) and future (2050) emissions to the North Sea and Baltic Sea from the spring barley production system studied.

While damage modelling may facilitate communication of (more understandable) results connected with the higher environmental relevance of longer pathway coverage (completeness), the loss of transparency and especially the additional uncertainties (parameter, model, or scenario) (Bare et al., 2000) may decrease the validity of the results in less robust models. Improving the DF modelling by means of spatially differentiated quantification of the fraction of species affected (as PAF) that potential becomes extinct (as PDF) in a spatial unit, may contribute to overcome the uncertainty of the model simplification that constitutes the DF. Such model improvements may embrace the inclusion of species vulnerability, uniqueness, ecosystem resilience, or functional diversity indicators – see e.g. Souza et al. (2013), Verones et al. (2015).

3.4. Implications for decision-makers and LCIA model developers

The inclusion of spatial differentiation is a valuable addition to any LCIA method, as long as there is a significant variability in the relevant parameters, not only to increase its discriminatory power (Udo de Haes et al., 1999), but also to add an extra information level to the decision-making process. Those who benefit from LCA results may adopt such differentiated information especially when dealing with human activities and emissions with local to regional impacts, like marine eutrophication. Complementary, it may provide useful analysis of supply chains with emissions at different locations with potentially differentiated impacts and increased the quality of the results produced in support of sustainability assessment.

The LCI models should preferably be supplied with spatially differentiated input data in order to maintain and explore that feature later in the characterisation. The LC-IMPACT method currently delivers CFs for the marine eutrophication indicator at country-to-LME as the highest spatial resolution, with 214 combinations of emitting country to receiving LME (Azevedo et al., 2013). LCIA developers may then aim at introducing temporal- (if relevant) along with spatial-differentiation in the models. In particular, the temporal variability in the marine eutrophication phenomenon at the intra-annual scale (months or seasons) may have a significant impact on the biological processes that compose the characterisation model, e.g. nutrients limitation, marine primary productivity, and species succession (Cosme et al., 2015) or species sensitivity (Cosme and Hauschild, 2016a). Its inclusion can be seen as future model improvement or research opportunity. The flexibility (or adaptability) of model parameterisations may further adopt archetypes that represent possible degrees of confidence or intensity of pressures. Notwithstanding, the value of flexible parameterisations seems essential in modelling future impacts beyond the adoption of timeframe perspectives (Hauschild et al., 2013). The calculation of NRs has also to match the time variation with corresponding data at the necessary spatial- and time-resolution, as also noted by Sleeswijk et al. (2008).

LCA scores aggregated at damage level can be relevant to decision-makers (e.g. managers, regulators) in the assessment of sustainability of activities and options, but also for ecosystems management and conservation. The misleading sense of certainty and comprehensiveness can however mine the confidence on its application (Bare et al., 2000). So, facing the merits and limitations of both midpoint and damage modelling steps, the use of both sets of results is suggested for a sound(er) interpretation and for the development of consistent methods across impact indicators.

Overall, the adjustment of the CF parameterisation is essential for the forecasting of LCIA results and its application in management plans for e.g. the agriculture and energy sectors, their regulation, and technological development (see other DPSIR Responses in Figure 1).

The application of the precautionary principle to the DPSIR approach (Figure 1) aims at showing that it is possible to anticipate impacts and act before the environment is affected. The concept, formalised in the UN 'Earth Summit' in 1992, ensures that by using indicators and impact assessment tools, (the magnitude of) the effects of future climatic changes can be already estimated and (some of) the Impacts anticipated, based on the present knowledge of the Pressures, so that Responses may be implemented sooner. In this line, specific LCIA indicators (such as marine eutrophication) are valuable contributions to support the precautionary approach, and so is the modelling of future impacts.

4. Conclusions

A novel characterisation model for nitrogen emissions from spring barley production was applied. The main improvement to the LCIA midpoint CFs is the inclusion of ecosystem exposure and effects to biota, by improving the commonly used 'increase in N concentration' in marine water to a 'fraction of species (as PAF) affected' by the eutrophication impacts in the marine coastal compartment. A first attempt to account for potential future climatic pressures, relevant to the marine eutrophication phenomenon, in a 2050 scenario was implemented, based on corresponding altered emission flows and modified parameterisation in the CF estimation.

Normalisation of results from present and future scenarios was compared, by estimating NRs based on total annual impacts (domestic inventory of background interventions), on ecological carrying capacity, and the newly proposed method. The comparison shows consistent results and also point to the value of adding spatial differentiation to the indicator's modelling framework.

The (i) inclusion of the time variation feature in CF modelling of marine eutrophication impacts, (ii) the characterisation of emissions at a spatially differentiated scale, and (iii) the identification of the need for adequate inventory data to assess future scenarios, constitute the main outcomes of the present study. Further research is needed to reduce the uncertainty of the parameterisation under future conditions extending the coverage of the climatic change aspects into the impact pathway and to tighten projections of future emissions.

The findings of this exploratory research point to the relevance of including time and spatial differentiation in characterisation modelling in LCIA. It also serves as a proof of concept that this kind of forecast modelling can, and should, be included in LCA. Finally, modelling the temporal variability of both inventory data and impacts appears central in exploiting the potential of LCA and fostering its legitimate application in decision support for scenario and precautionary analyses.

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