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Review article

Towards a meaningful assessment of marine ecological impacts in life cycle assessment (LCA)



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ABSTRACT

Human demands on marine resources and space are currently unprecedented and concerns are rising over observed declines in marine biodiversity. A quantitative understanding of the impact of industrial activities on the marine environment is thus essential. Life cycle assessment (LCA) is a widely applied method for quantifying the environmental impact of products and processes. LCA was originally developed to assess the impacts of land-based industries on mainly terrestrial and freshwater ecosystems. As such, impact indicators for major drivers of marine biodiversity loss are currently lacking. We review quantitative approaches for cause–effect assessment of seven major drivers of marine biodiversity loss: climate change, ocean acidification, eutrophication-induced hypoxia, seabed damage, overexploitation of biotic resources, invasive species and marine plastic debris. Our review shows that impact indicators can be developed for all identified drivers, albeit at different levels of coverage of cause–effect pathways and variable levels of uncertainty and spatial coverage. Modeling approaches to predict the spatial distribution and intensity of human-driven interventions in the marine environment are relatively well-established and can be employed to develop spatially-explicit LCA fate factors. Modeling approaches to quantify the effects of these interventions on marine biodiversity are less well-developed. We highlight specific research challenges to facilitate a coherent incorporation of marine biodiversity loss in LCA, thereby making LCA a more comprehensive and robust environmental impact assessment tool. Research challenges of particular importance include i) incorporation of the non-linear behavior of global circulation models (GCMs) within an LCA framework and ii) improving spatial differentiation, especially the representation of coastal regions in GCMs and ocean-carbon cycle models.

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1. Introduction

Human demands on marine resources and space are currently unprecedented and are expected to further increase in the near future (Millennium Ecosystem Assessment, 2005; Foley et al., 2010). Seen by many (e.g. the European Commission, 2012, and the Australian Government, 2015) as the next frontier for economic development, new marine activities, such as renewable energy harvesting, carbon sequestration and gas hydrate mining, are rapidly emerging (Millennium Ecosystem Assessment, 2005). Traditional marine activities remain important as well. Offshore oil and gas production, world sea trade and marine aquaculture are projected to continue growing in importance over the coming decades (OECD, 2010; US Energy Information Administration, 2010; FAO, 2014). Marine capture fisheries have stabilized but remain important, given that fish consumption per capita is increasing (FAO, 2014; Thrane et al., 2009). Additionally, several marine industrial activities have been expanding from coastal, shallow waters into progressively deeper waters (e.g. ultra-deep oil production fields in Brazil and deep-sea fisheries in the Northwest Atlantic) and to previously unexplored areas (e.g. the Arctic lower shelf) (Millennium Ecosystem Assessment, 2005; US Energy Information Administration, 2010; Crowder et al., 2008; Dalsøren et al., 2007; Morato et al., 2006).

Concurrently, observed declines in marine biodiversity have been attributed to human activities (Millennium Ecosystem Assessment, 2005; Foley et al., 2010; Pauly et al., 2002; Costello et al., 2010). Two comprehensive studies, the Millennium Ecosystem Assessment (2005) and CenSus of Marine Life (Census) (Costello et al., 2010), have documented declines in marine biodiversity on a global scale. Together, these studies identified climate change, ocean acidification, eutrophication-induced hypoxia, habitat change (including seabed damage), overexploitation and invasive species as main drivers of marine biodiversity loss (Millennium Ecosystem Assessment, 2005; Costello et al., 2010). Another prominent driver, albeit currently poorly understood, in marine environments arises from plastic debris (Gall and Thompson, 2015). While some drivers, such as climate change and eutrophication-induced hypoxia, can be traced back to both land-based and sea-based activities; other drivers, such as habitat change, invasive species, and overexploitation, are predominantly linked to sea-based activities (Millennium Ecosystem Assessment, 2005; Foley et al., 2010; Costello et al., 2010; Halpern et al., 2008). A quantitative understanding of the environmental impact of these industrial activities on the marine environment would greatly improve the robustness and completeness of life cycle assessment (LCA).

LCA is a standardized method to evaluate the environmental impact of a product or process over its full life cycle. In LCA, life-cycle inventory data, including emissions produced (e.g. kg CO₂) and resources used (e.g. m³ of water consumed) are converted to impact scores for various environmental categories, such as global warming, human toxicity, and aquatic eutrophication. Potential inventory data for seven major drivers of marine biodiversity loss are included in Table 1. LCA uses stressor-specific and impact-category specific characterization factors (CF) to convert inventory data to potential impacts. These CFs express the fate and effect of a stressor per unit of intervention (emission or used resource). The fate factor models the spatial distribution and intensity of

a unit intervention and is generally obtained from environmental fate models (Curran et al., 2011; Huijbregts et al., 2011). The effect factor relates the intensity of an intervention to a quantified effect, such as the potentially disappeared fraction (PDF) of species (Curran et al., 2011).

LCA is particularly well-suited to identify potential trade-offs that occur across impact categories or life-cycle stages and consequently offers the possibility to optimize the overall environmental performance of a product or process. These key advantages have resulted in an increased LCA application to various technologies in the last decade (Guinée et al., 2011), including several sea-based technologies, such as offshore oil and gas production (Veltman et al., 2011), offshore wind turbines (Weinzettel et al., 2009), marine capture fisheries (Avadí and Fréon, 2013; Pelletier et al., 2007; Ziegler and Valentinsson, 2008) and marine aquaculture (Aubin et al., 2009). LCA, however, was originally developed to assess the impact of land-based product systems on mainly terrestrial and freshwater ecosystems and currently lacks a marine impact focus. The standard suite of LCA impact categories contains only two that are relevant for the marine environment, namely marine eutrophication and marine ecotoxicity (ReCiPe, 2009). Relevant impact indicators for other important drivers of marine biodiversity loss, such as climate change, ocean acidification, eutrophication-induced hypoxia, overexploitation of fishery resources, invasive species and habitat change, are yet lacking. For an adequate assessment of the environmental performance of coastal and offshore industrial activities in LCA it is of utmost importance to develop indicators for important drivers of marine biodiversity loss.

Here, we review quantitative approaches for the environmental assessment of seven major drivers of marine biodiversity loss, i.e. climate change, ocean acidification, eutrophication-induced hypoxia, seabed damage, invasive species, overexploitation and marine plastic debris. We also provide recommendations on how to quantitatively incorporate these drivers of marine biodiversity loss in LCA. We focus on quantitative approaches that have been used to assess marine impacts on a global and/or regional scale, as LCA requires cause–effect models that can be consistently applied to various geographic regions, and impact indicators that are comparable across ecosystems. For each driver, we review the state-of-the-art of cause–effect modeling in terms of taxonomic and geographic coverage and the termination point of cause–effect modeling, i.e. whether the approach quantifies impacts on biological systems, consistent with LCA effect (endpoint) modeling, or a change in environmental state, consistent with LCA fate (midpoint) modeling. We identify main current (conceptual) limitations and provide recommendations on steps that need to be taken in order to incorporate the studied drivers of marine biodiversity loss in LCA.

2. Climate change

2.1. Cause–effect

“Climate change” encompasses a range of physical and chemical modifications to the ocean, primarily resulting from greenhouse gas (GHG) driven global warming. Well-documented changes include: i) an increase in average sea surface temperature (SST) and in the frequency and intensity of SST anomalies, ii) a rise in average global sea

level and iii) sea-ice retreat in the Arctic and the West Antarctic Peninsula (Hoegh-Guldberg and Bruno, 2010; Doney, 2010). Furthermore, climate change is predicted to result in a loss of dissolved oxygen in the ocean interior (Keeling et al., 2010) and has been linked to observed changes in upper-ocean salinity (Doney et al., 2012). These modifications occur globally, but vary spatially in intensity owing to ocean circulation, wind patterns and interaction with natural modes of climate variability (e.g. El Niño/Southern Oscillation and North Atlantic Oscillation) (Doney et al., 2012; Gruber, 2011).

Marine organisms have a discrete range of abiotic conditions under which physiological function is optimized and thresholds where performance is compromised and lost (Hofmann and Todgham, 2009). The breadth of this range is highly variable between species. Large-scale changes in ocean abiotic conditions therefore have direct effects on the physiological performance (e.g. growth and reproduction) and behavior of marine organisms, which can propagate to population- and community-level changes (Hoegh-Guldberg and Bruno, 2010; Doney et al., 2012). For many marine species it is thought to be unlikely that they can keep pace with changing local conditions through acclimatization (an adjustment of physiology within individuals), adaptation (increased abundance and reproduction of tolerant genotypes over generation) or phenotypic plasticity (e.g. change in timing of annual events) (Doney et al., 2012; Parmesan, 2006; Somero, 2011). Migration is necessary if individuals and/or populations are to remain in their climatic zone of preference. A failure to migrate can lead to higher mortality, reduced growth and reproduction rates, and at the extreme, to local extinction (Doney et al., 2012; Somero, 2011). Migration leads to changes in species distribution and abundance, and potentially disrupts community interactions (Perry et al., 2005; Pörtner and Knust, 2007).

The biological impacts of climate change are currently best understood and documented for ocean warming. Organism-level adverse physiological effects due to temperature increase have been mechanistically linked to a loss of aerobic scope (Pörtner and Knust, 2007) and frequently demonstrated in laboratory studies (Somero and DeVries, 1967; Somero, 2010; Sorte et al., 2011; Peck, 1989; Peck et al., 2004; Urban, 1994). Regional-level field studies in the German Wadden Sea and Tasmania, have shown a reduced growth and abundance of marine fish species when sea water temperature surpassed the threshold for optimal organism performance (Pörtner and Knust, 2007; Neuheimer et al., 2011). Furthermore, meta-analyses have documented pole-ward range shifts of hundreds of kilometers in a few decades across many species and geographic regions (Perry et al., 2005; Beaugrand et al., 2002; Gregory et al., 2009; Richardson and Poloczanska, 2008; Nye et al., 2009). Presently, one of the most profound and clear effects of climate change on the world's ocean is the observed increase in mass-bleaching and mortality of habitat-forming coral reefs, due to an increase in intensity and frequency of thermal anomalies (Hoegh-Guldberg and Bruno, 2010; Liu et al., 2003; Hoegh-Guldberg et al., 2007).

2.2. Quantitative approaches and LCA perspective

Cause-effect models have been developed for two distinct impacts of climate change: degradation and loss of habitat-forming coral reefs and changes in distribution of marine species.

Donner et al. (2005) and Donner (2009) projected future degradation of global coral reefs due to thermal stress by coupling atmosphere-ocean general circulation models (GCM) with a model for coral bleaching. This model is based on the observed relationship between mass coral bleaching and the annual accumulation of degree heating months (DHM; a metric that incorporates both the magnitude of SST spikes, i.e. °C above the maximum monthly mean, and the number of months with a SST spike) (Liu et al., 2003). Globally-averaged impacts are expressed as the bleaching frequency per decade or as the number of bleaching months per year. This modeling approach provides an illustration of the globally averaged potential impact of future

climate change on warm-water coral reefs. Variation in model projections of coral bleaching is predominantly caused by inter-model variability of GCMs (Donner et al., 2005). At present, the approach has two main limitations: i) most current climate models have a rather coarse spatial resolution, i.e. a single grid cell may span the entire continental shelf width, and are better suited to represent the mean temperature of an area of ocean containing coral reefs than the temperature surrounding an individual coral reef (Donner et al., 2009; Stock et al., 2011); and ii) presently a fixed bleaching threshold is used, although bleaching susceptibility is shown to vary across coral taxa as well as across coral-symbiont couples (Donner et al., 2009; Carilli et al., 2012). Improvements in GCMs, particularly a higher grid resolution in the tropics, higher vertical resolution in the upper ocean, and site-specific hydrodynamic models, and the use of variable bleaching thresholds are crucial in refining projections for individual reefs (Donner et al., 2009; Carilli et al., 2012). However, these improvements are thought to have a small effect on the globally averaged prognosis of coral bleaching (Donner et al., 2005; Donner et al., 2009). Coupling of atmosphere-ocean GCMs with a model for coral bleaching provides an opportunity to develop CFs for loss of habitat-forming coral reefs expressed as potential habitat loss per unit of GHG emission ($\text{km}^2_{\text{habitat loss}} \text{day kg}_{\text{GHG}}^{-1}$).

Several studies have coupled GCMs with bioclimatic envelope models (ecological niche models) to project future distributions of marine species under climate change induced modifications of oceanic conditions. Impacts are expressed as species range shift, habitat loss and/or species loss rate. The most extensive study includes 1066 commercially exploited cold-blooded marine species and covers the global ocean, although data coverage is low for high-latitude regions (e.g. northern polar areas, Hudson Bay and Antarctica) (Stock et al., 2011; Cheung et al., 2009). The bioclimatic envelope model approach has also been used to quantify potential changes in the global distribution of 115 marine mammals (Kaschner et al., 2011). Predictions for marine mammals are currently more uncertain than for cold-blooded organisms (Kaschner et al., 2011), in part, due to restricted data availability on species occurrence and habitat usage (Kaschner et al., 2011; Tyberghein et al., 2012). Additionally, marine mammals are less constrained by physical environmental conditions than cold-blooded species: the distribution and density of food supply is an important factor in defining their distribution. However, the distribution and density of prey is not explicitly considered in the bioclimatic envelope model approach (Kaschner et al., 2011; Robinson et al., 2011). Projections of species range shifts are also sensitive to inter-model variability of GCMs (Stock et al., 2011; Cheung et al., 2009). At present, bioclimatic envelope model predictions are considered to be inaccurate for individual species at a regional scale, particularly in coastal regions (Stock et al., 2011; Cheung et al., 2009). This is due to the coarse spatial resolution of most GCMs (Stock et al., 2011) and due to limitations in the biological niche modeling approach, particularly the limited consideration of fundamental biotic interactions, such as feeding and competition (Stock et al., 2011; Cheung et al., 2009). Coupled GCM-biological niche models can, however, provide a prediction of globally averaged species range shifts, as uncertainties in physical and biological data important for local scale predictions are largely overcome by the large sample size and taxonomic and geographic coverage (Gregory et al., 2009; Cheung et al., 2009). Also, model evaluation with field data from the North Sea showed that predictions of current species range shifts agree reasonably well with observations (Cheung et al., 2009). Coupling GCMs with biological niche models provides an opportunity to develop CFs for species range shifts expressed in terms of latitudinal range shift per unit of GHG emission ($\text{km}_{\text{shifted}} \text{day kg}_{\text{GHG}}^{-1}$).

There are a few issues that need to be addressed in the development of these CFs:

- i) GCMs are non-linear (IPPC, 2014) whilst LCA currently derives fate factors mostly from linear environmental fate models. It should be tested whether this non-linearity is relevant in terms

Table 1
Summary of approaches for quantifying the effect of seven stressors of the marine environment on biodiversity.

Inventory data	Developmental stage	Indicator	Unit	Biodiversity measure	Taxonomic coverage	Geographic coverage	Reference(s)
Climate change kg GHG emissions	Quantitative technique	Warm water coral reef degradation	$\text{km}^2_{\text{habitat loss}} \text{day kg}_{\text{GHG}}^{-1}$	Area habitat loss	Warm water coral reefs	Regional	Donner et al. (2005), Donner (2009)
kg GHG emissions	Quantitative technique	Globally averaged latitudinal species range shifts	$\text{km}_{\text{shifted}} \text{day kg}_{\text{GHG}}^{-1}$	Species range shifts	1066 commercially exploited cold-blooded marine species	Global	Stock et al. (2011), Cheung et al. (2009)
Ocean acidification kg CO ₂ emissions	Quantitative technique	PAF of pH sensitive species globally	PAF day kg CO_2^{-1}	Potentially affected fraction of species	Calcifying species sensitive to pH change (up to 40 experiments per SSD)	Global	Zeebe et al. (2008), Doney et al. (2009), Azevedo et al. (2015)
Eutrophication-induced hypoxia kg N emissions: NO _x and NH ₃ to air and N to surface freshwater, to groundwater and directly to marine coastal waters.	LCA	PAF of benthic species within N-impacted LME(s)	PAF $\text{m}^3 \text{year kg N}^{-1}$	Potentially affected fraction of species	57 species of fish, crustaceans, molluscs, echinoderms, annelids or cnidarians	Regional	Cosme (in preparation), Cosme et al. (2015), Cosme and Hauschild (submitted for publication)
Seabed damage kg caught fish, including bycatch and discards	Quantitative technique	Habitat degradation, based on average area trawled per unit catch and habitat specific recovery time	$\text{km}^2 \text{year}^{-1} \text{kg}_{\text{caught fish}}^{-1}$	Area habitat degraded	–	Regional	Foden et al., (2010)
kg particles and km ² of distribution	Quantitative technique	Volume- and time-integrated PAF	PAF $\text{m}^3 \text{day kg}_{\text{particle}}^{-1}$	Potentially affected fraction of species	Up to 32 species	Regional	Veltman et al. (2011)
(Over)exploitation kg caught fish, including bycatch and discards	LCA	Maximum potential regeneration time	$\text{year kg}_{\text{caught fish}}^{-1}$	Perturbation to single-species stock(s)	138 commercial stocks worldwide	Stock-scale	Langlois et al. (2014)
kg caught fish, including bycatch and discards	LCA	Biomass regeneration time at the ecosystem level	$\text{year kg}_{\text{caught fish}}^{-1}$	Loss of energy available to support biodiversity	–	Regional	Langlois et al. (2014)
kg caught fish, including bycatch and discards	LCA	Lost potential yield (LPY) averaged over a 20, 30 or 100 year time horizon.	$\text{kg}_{\text{LPY}} \text{kg}_{\text{caught fish}}^{-1} \text{year}^{-1}$	Perturbation to single-species stock(s)	31 major European stocks	Stock-scale	Emanuelsson et al. (2014)
kg caught fish, including bycatch and discards	Quantitative technique	Projected changes in functional group biomass coupled with knowledge on intrinsic vulnerable of species to fishing: the Depletion Index (DI)	DI $\text{kg}_{\text{caught fish}} \text{year}^{-1}$	Change in species composition and mean abundance.	39 functional groups Knowledge of intrinsic vulnerability available for 733 fish species.	Regional	Alder et al. (2007)
Invasive species Transport by sea – km traveled or kg transported	Theoretical	PDF	PDF $\text{m}^3 \text{year kg}^{-1}$	Species richness loss	–	Regional	Hanafiah et al. (2013)
Marine plastic debris No methods for quantifying the effect of plastic waste on biodiversity at scales greater than individual organisms have yet been proposed.							

- of a marginal emission increment on the present GHG emissions.
- ii) Individual GHGs differ in their warming potential, due to their different radiative properties and lifetimes in the atmosphere (IPPC, 2014). This is usually addressed by expressing the emitted amount of long-lived GHGs as CO₂-equivalent emissions that would cause the same time-integrated radiative forcing over a given time horizon (IPPC, 2014). Furthermore, climate change induced ocean warming is characterized by a lag-time, due to the large thermal inertia of the oceans. Both issues require selection of a relevant time horizon.
 - iii) Impacts on coral reefs are based on a fixed threshold value for coral bleaching. Thresholds can be problematic for LCA as effect factors are usually calculated based on the derivative of a continuous dose–response function (e.g. Huijbregts et al., 2011).

These issues could be addressed by calculating CFs based on a simultaneous integration of fate and effect over time based on present-day GHG emissions and a well-defined time horizon. To address uncertainty in GCM predictions, a multi-model mean output of various GCMs with contrasting uncertainties should be used (Stock et al., 2011). Further research should focus on refinement of both impact assessment approaches, particularly in improving GCM skills for coastal regions. Additionally, there is a need to further develop the GCM-biotic niche modeling approach for a larger number of taxonomic groups, particularly for polar species and marine mammals.

3. Ocean acidification

3.1. Cause–effect

Ocean acidification is the shift in seawater carbonate chemistry and the reduction in ocean surface pH, as a result of oceanic uptake of carbon dioxide (CO₂) (Zeebe et al., 2008; Doney et al., 2009). The ocean is the primary carbon sink on centennial to millennial timescales (Kheshgi, 2004) and has taken up approximately 40% of anthropogenic CO₂ emissions over the past 200 years (Zeebe et al., 2008). This has resulted in: i) an alteration of seawater carbonate chemistry by shifting the dissolved inorganic carbon equilibrium away from carbonate (CO₃²⁻) towards more bicarbonate (HCO₃⁻) and more CO₂ (Zeebe et al., 2008; Doney et al., 2009), ii) a global reduction in average ocean surface pH (Orr et al., 2005), and iii) shoaling of saturation horizons of carbonate minerals, such as aragonite (Ω_{arag}) and calcite (Ω_{Ca}), in the Atlantic, Pacific and Indian Oceans (Feely et al., 2004). These processes are well verified from hydrographic surveys and time series data (Zeebe et al., 2008; Doney et al., 2009) and occur on a global scale, albeit varying geographically in intensity (Doney et al., 2009).

Such large scale changes in seawater carbonate chemistry are expected to have adverse effects on individual organisms, communities and ecosystems (Doney et al., 2009; Fabry et al., 2008). Negative effects are expected for calcifying organisms, such as corals, echinoderms, crustaceans and mollusks, because ocean acidification reduces their ability to produce and maintain shells or other structures of calcium carbonate (CaCO₃) (Orr et al., 2005; Fabry et al., 2008; Kleypas and Yates, 2009). The majority of marine calcifiers tested to date have shown declines in net calcification rates in laboratory and mesocosm studies in response to changing carbonate saturation levels (Guinotte and Fabry, 2008; Kroeker et al., 2010; Pandolfi et al., 2011).

Calcification responses are, however, highly variable and often non-linear (Kroeker et al., 2010; Pandolfi et al., 2011; Wernberg et al., 2012). Among the main reasons for this are morphological differences leading to differences in species sensitivity (Kroeker et al., 2010; Pandolfi et al., 2011; Fabry, 2008; Kroeker et al., 2011), and the potential effect of ocean acidification on a variety of physiological processes, including calcification, acid–base (metabolic) physiology, and photosynthesis, which are correlated to different components of the carbonate system.

(Hofmann and Todgham, 2009; Doney et al., 2009; Kleypas and Yates, 2009; Pörtner, 2008; Veron, 2011; Andersson and MacKenzie, 2012). Moreover, differences in experimental conditions may lead to variable responses (Pandolfi et al., 2011; Fabry, 2008).

Counter-intuitive results are also found: several studies have shown increased calcification rates under reduced pH (Wood et al., 2008; Findlay et al., 2009). This has been attributed to up-regulation of calcification rates at the expense of other physiological processes, such as metabolic rates (Findlay et al., 2009). Also, ocean acidification has been shown to positively affect carbon fixation rates in some calcifying photosynthetic organisms (Kroeker et al., 2010; Pandolfi et al., 2011; Ries et al., 2009).

While it is apparent that changing seawater chemistry will have serious negative consequences for most marine calcifiers, a strong (mechanistic) cause–effect understanding of how changes in ocean carbonate chemistry affect organism fitness and survival, which (physiological) response variables are most sensitive and which dissolved inorganic carbon specie(s) are most important to marine organisms is yet lacking (Pörtner, 2008; Andersson and MacKenzie, 2012). Additionally, potential trade-offs among calcification and other physiological responses, and the interaction between calcification and photosynthesis, are currently poorly understood (Hofmann and Todgham, 2009; Ries et al., 2009).

Most of the work on ocean acidification impact assessment has so far concentrated on calcifying organisms, particularly on warm-water corals (Kroeker et al., 2010; Veron, 2011). Consequently, it is even less well understood how non-calcifying organisms, communities and ecosystems, will respond to ocean acidification (Fabry et al., 2008; Guinotte and Fabry, 2008; Kroeker et al., 2010; Pandolfi et al., 2011). In laboratory studies, changes in carbonate chemistry had a positive effect on growth of several non-calcifying organisms, including fish, fleshy algae and crustaceans (Kroeker et al., 2010). Limited field studies in temperate and tropical natural shallow water CO₂ vents indicate that ocean acidification reduces the occurrence of calcifying organisms, but increases the occurrence of fleshy macro-algae and sea-grasses (Kroeker et al., 2011; Fabricius et al., 2011; Hall-Spencer et al., 2008). Community and ecosystem-level effects include a decrease in species diversity and biomass, and disruption of trophic structure due to shifts in predator–prey and other competitive interactions between taxa (Kroeker et al., 2011; Fabricius et al., 2011). The response of individual organisms, populations and communities to continuous long-term exposure to lower, and more gradual changes in pH, or the capacity of these organisms to acclimatize or adapt, is largely unknown (Hofmann and Todgham, 2009; Doney et al., 2009; Fabry et al., 2008).

3.2. Quantitative approaches and LCA perspective

Several studies have quantified global changes in surface ocean pH and carbonate mineral saturation state (often expressed as aragonite saturation state) based on ocean carbon-cycle models (Zeebe et al., 2008; Orr et al., 2005; Caldeira and Wickett, 2003). The chemistry of CO₂ in seawater is well-known and ocean carbon-cycle models can fairly accurately predict changes in carbonate chemistry as a function of changing atmospheric CO₂ concentrations (Zeebe et al., 2008; Doney et al., 2009). These quantitative approaches are spatially-explicit, cover the global ocean and are consistent with LCA fate modeling. A fate factor could be developed that describes the change in ocean carbonate chemistry per unit CO₂ emission (e.g. $\Delta\text{pH day kg CO}_2^{-1}$ or $\Delta\Omega_{\text{arag}} \text{ day kg CO}_2^{-1}$).

A major limitation of ocean-carbon cycle models is that coastal regions are presently not well represented (Andersson and MacKenzie, 2012). Coastal environments are characterized by much larger fluctuations in seawater carbonate chemistry than the open ocean, due to tidal dynamics, coastal upwelling, and major fluvial inputs (Andersson and MacKenzie, 2012), and evaporation and precipitation in shallow tropical basins (Hu et al., 2015). These near-shore circulation processes

are currently not well represented in ocean-carbon cycle models (Hoegh-Guldberg et al., 2007; Fabry et al., 2008; Feely et al., 2008). Improvements are essential as a significant proportion of benthic calcifying organisms (e.g. mussels, oysters, corals, coralline algae) reside in shallow water coastal environments. In addition, in many coastal regions, seawater $p\text{CO}_2$ is already significantly higher, and pH lower, than expected from CO_2 equilibrium partitioning between the atmosphere and the open ocean, in part due to biological processes (Andersson and MacKenzie, 2012).

A method for quantifying the effect of ocean acidification on marine biodiversity has been developed by Azevedo et al. (2015). The method utilizes a species sensitivity distribution (SSD) approach to quantify the effects of decreasing pH on growth, reproduction and survival of calcifying organisms. Whilst taking an SSD approach avoids the necessity of a mechanistic understanding of the effects of ocean acidification, more studies assessing species responses to ocean acidification are required to facilitate the development of a more robust effect factor. The SSDs are currently spatially-generic due to limited species-response data. Standardization of protocols and data reporting guidelines for carbonate system manipulation and calcification experiments (Doney et al., 2009) would help accelerate availability of data suitable for inclusion in the construction of SSDs. In addition, the effect factor only reflects the response of calcifying species sensitive to pH change. A broader range of less sensitive species need to be tested, particularly from polar and tropical regions, as well as primary producers, pelagic invertebrates and fish (Wernberg et al., 2012). Greater data availability would enable spatial differentiation of SSDs, for example by climatic zone. The fate factor $\Delta\text{pH day kg CO}_2^{-1}$, calculated at a global scale to correspond with the spatially-generic effect factor, could then potentially be coupled with the results of the SSD curves. Using an average approach, the potentially affected fraction (PAF) of calcifying species sensitive to pH change could then be estimated, and together with the fate factor provide a CF with the unit $\text{PAF day kg CO}_2^{-1}$.

4. Eutrophication-induced hypoxia

4.1. Cause-effect

Eutrophication-induced hypoxia is a predominant driver for biodiversity loss in coastal ecosystems world-wide (Millennium Ecosystem Assessment, 2005; Costello et al., 2010). Although hypoxia can occur naturally in coastal environments, evidence is accumulating that anthropogenic factors are driving expansion in the duration, intensity and extent of low oxygen zones world-wide (Doney, 2010; Levin et al., 2009; Howarth et al., 2011). At present, anthropogenic factors are responsible for virtually all hypoxia in estuaries and on inner continental shelves and have worsened dissolved oxygen conditions in many fjords (Rabalais et al., 2010). One of the largest areas of hypoxia in the world is located in the Gulf of Mexico (Rabalais et al., 2002).

The most direct human cause of hypoxia is eutrophication (Middelburg and Levin, 2009), i.e. the excessive input of nutrients, specifically reactive nitrogen (e.g. NO_x , NH_3 , NO_3^-) in the ocean, and reactive nitrogen and phosphorus in estuaries and continental shelf water (Correll, 1998). The enhanced input of nutrients stimulates excessive phytoplankton growth, which leads to a greater production of particulate organic matter (POM) in the water column or on the seabed. Near to river mouths, inflows of allochthonous material contribute to the total load of POM (Bianchi et al., 2010). When the resulting organic matter exceeds the capacity of metazoan consumers to metabolize it, the remaining organic matter settles to the density barrier between surface waters and deep ocean water or the seabed, where it is decomposed, mainly by heterotrophic bacteria in a process that uses up oxygen (Levin et al., 2009). If oxygen is not supplied by advective processes, diffusion and/or photosynthetic production, then the bottom waters can become hypoxic or, in extreme cases, anoxic (Rabalais et al., 2010; Gray et al., 2002).

The physical structure of the system is a key factor in the formation of hypoxic water masses (Howarth et al., 2011). Hypoxia is usually associated with a density barrier, caused by temperature, salinity or both, that impedes oxygen supply through diffusion (Rabalais et al., 2010). Additionally, hypoxic water masses are more likely to occur in marine systems with long water residence times and low tidal flush rates (Howarth et al., 2011; Rabalais et al., 2010; Diaz and Rosenberg, 2008).

Hypoxia primarily affects the bottom layer of the water column. Benthic fauna and bottom-dwelling fishes are consequently more severely impacted than species inhabiting the upper water column (Breitburg et al., 2009). The effect of hypoxia on sessile benthic organisms is often lethal (Vaquer-Sunyer and Duarte, 2008). For more mobile benthic organisms, effects are often sub-lethal and include a reduction in growth and reproduction, physiological stress and a forced migration to more suitable habitats (Vaquer-Sunyer and Duarte, 2008). Hypoxia effects increase with an increased intensity, duration and frequency of low oxygen zones and with co-occurrence of other stressors, particularly hydrogen sulfide and elevated temperature (Levin et al., 2009).

Organism-level adverse effects are well-documented from laboratory experimental studies for a range of benthic organisms, including mollusks, crustaceans, and fish (Gray et al., 2002; Vaquer-Sunyer and Duarte, 2008). Field studies in well-known hypoxic zones, such as Tokyo Bay, the northern Gulf of Mexico, and the western Baltic Sea, have shown that initial community level effects include an altered benthic species composition and a reduction in their diversity (Wu, 2002; Kodama and Horiguchi, 2011). After abatement of hypoxia, recovery of the impacted benthic area can occur, either through migration or larval settlement, with the recolonization time depending on the frequency and intensity of hypoxia events (Levin et al., 2009; Kodama and Horiguchi, 2011; Botter-Carvalho et al., 2011). Long-term hypoxic periods, which typically occur once per year and last for several months, and repeated exposure, occurring more frequently than seasonally with each event lasting days-weeks, can result in severe changes in benthic communities with a shift in composition favoring those species with opportunistic life history strategies (Diaz and Rosenberg, 2008; Diaz and Rosenberg, 1995; Conley et al., 2007; Essington and Paulsen, 2010). In extreme cases, long-lasting events of anoxic conditions, so called dead zones, are created that support little or no marine life (Rabalais et al., 2002; Diaz and Rosenberg, 2008).

4.2. Quantitative approaches and LCA perspective

A spatially-explicit LCA method will be available to quantify the impacts of eutrophication-induced hypoxia in coastal ecosystems, more specifically the 66 spatial units of continental shelves represented by the Large Marine Ecosystem (LME) biogeographical classification system. The characterization factors [$\text{PAF m}^3 \text{ year kg}^{-1}$] for marine eutrophication-induced hypoxia impacts due to nitrogen (N) emissions to water, air and agricultural soils, are spatially differentiated by country and LME unit.

These endpoint characterization factors are constructed from fate (Cosme, in preparation), exposure (Cosme et al., 2015) and effect factors (Cosme & Hauschild, submitted for publication). The fate factor models the nitrogen load within a large marine ecosystem and is a function of the fraction of N emissions exported from a country to adjacent LME receiving cells and the N-loss rate in the receiving LME cell(s). Spatially-explicit exposure factors [$\text{kg O}_2 \text{ kg N}^{-1}$] provide a nitrogen to oxygen consumption conversion potential based on carbon flux processes (Ducklow et al., 2001) and bacterial degradation (Del Giorgio and Cole, 1998; Iversen and Ploug, 2010). The impact of reduced dissolved oxygen concentration on biota is estimated by the effect factor (PAF) [$\text{m}^3 \text{ kg O}_2$]. Effect factors represent the average change in effect (ΔPAF , including sub-lethal effects at behavioral and physiological level) due to a decrease in dissolved oxygen availability and are based on SSDs that include the response of up to 57 species of fish, crustaceans, mollusks, echinoderms, annelids or cnidarians. Effect factors are

available at a 5 climatic zone resolution because species sensitivity data are too few to enable spatial differentiation to the resolution of 66 LMEs.

The method developed by Cosme and Hauschild (Cosme, in preparation; Cosme et al., 2015; Cosme & Hauschild, submitted for publication) is a useful development for inclusion of impacts on the marine environment from eutrophication-induced hypoxia in LCA. Further research efforts should focus on incorporating spatial differentiation with respect to limiting nutrient (i.e. N or P); for example consideration of the contribution of reactive phosphorus to eutrophication in coastal waters. In addition, delayed recovery of communities following cessation of hypoxic conditions has been observed (Lim et al., 2006). As successional patterns are not well understood, impacts associated with delayed recovery from anoxic conditions, analogous to transformation impacts in the land use impact category (Milà et al., 2007), are not currently accounted for.

5. Seabed damage

5.1. Cause–effect

Seabed damage is one of the predominant drivers for biodiversity loss in coastal ecosystems world-wide (Millennium Ecosystem Assessment, 2005). Disruptive fishing techniques, including bottom trawling, are considered among the major causes of physical destruction of marine coastal habitats at global scales (Airoldi and Beck, 2007). In 2002 it was estimated that approximately 75% of the continental shelf area worldwide is trawled every year (Kaiser et al., 2002). On regional scales, extraction due to dredging activities and smothering (or physical burial) due to offshore oil and gas production are also important (Eastwood et al., 2007; Foden et al., 2010). Physical disturbances can be characterized in terms of pressure: abrasion (e.g. bottom trawling), smothering (e.g. dredged material disposal, cutting pile formation due to offshore oil and gas production) and extraction (e.g. aggregate dredging) (Eastwood et al., 2007).

The overall impact of physical disturbances on benthic communities is a function of three interacting processes: disturbance characteristics (scale and intensity), initial benthic response and recovery time (Thrush et al., 1996; Thrush et al., 2008; Bolam and Rees, 2003; Kaiser et al., 2006). The primary response of benthic organisms to a physical disturbance is often severe damage or mortality (Bolam and Rees, 2003; Kaiser et al., 2006; Collie et al., 2000). Particularly impacted are sedentary organisms and filter feeders, such as anemones, soft corals and bivalve species (Kaiser et al., 2002; Widdicombe and Austen, 2001; Thrush and Dayton, 2002; Smit et al., 2008). This can result in a number of changes in benthic communities, including loss of habitat-structuring species, substantial decline in taxon abundance, and changes in species richness (Kaiser et al., 2002; Thrush and Dayton, 2002; Hiddink et al., 2006). After cessation of the physical pressure, the benthic community can recover (Kaiser et al., 2006). The recovery time is strongly related to environmental characteristics (Bolam and Rees, 2003; Collie et al., 2000). Structurally complex habitats (e.g. biogenic reefs) and relatively unstressed marine environments (deep, physically stable habitats experiencing infrequent sediment movement from wave-action) take substantially longer to recover from a physical disturbance than more naturally stressed areas (shallow estuarine habitats, which are physically transient), because species in naturally stressed environments are more adapted to unpredictable conditions and physical recovery is generally faster (Bolam and Rees, 2003; Collie et al., 2000; Hiddink et al., 2006; Foden et al., 2009). These environment-specific differences in recovery time are well-documented in meta-analyses of benthic ecosystem recovery from several physical pressures, including fishing, aggregate extraction, and dredged material deposition (Bolam and Rees, 2003; Kaiser et al., 2006; Foden et al., 2009). The recovery time furthermore depends on the scale of disturbance (Thrush

et al., 2008) and the number of successional stages required regaining the original community composition (Bolam and Rees, 2003).

Physical disturbances of the seabed can result in permanent community changes when the frequency and extent of disturbance outstrips the recovery potential (Thrush et al., 2008). Permanent community shifts from dominance by high biomass organisms, including bioturbating macrofauna, towards small-bodied opportunistic species, such as polychaetes, have been documented for heavily trawled areas, such as the Gulf of Mexico and the North Sea (Crowder et al., 2008; Jennings et al., 2001a; Jennings et al., 2001b).

5.2. Quantitative approaches and LCA perspective

Spatially-explicit quantitative impact assessment approaches have been developed for several physical pressures, including abrasion (e.g. bottom trawling), smothering (e.g. cutting pile formation), and extraction (e.g. aggregate extraction). Although targeting different pressures, impacts are commonly quantified by linking the distribution and intensity of a pressure to a quantitative measure of ecosystem sensitivity. Recovery time of an ecosystem is often used as a proxy for ecosystem sensitivity. However, recovery time is influenced by value choices, particularly the definition of recovery and choice of component(s) to indicate recovery.

Nilsson and Ziegler (2007) evaluated the impact of bottom-trawling on the seabed along the Swedish west coast in terms of the total seabed area persistently within a disturbance cycle i.e. the interval between trawling events was shorter than recovery time. Habitat-specific recovery times were used as a proxy for habitat sensitivity. These recovery times were obtained from the MarLIN database, which provides a systematic ranking of habitat recoverability as a result of physical disturbances per habitat type (MarLIN, 2015) and defines recovery of a habitat as the re-growth, re-colonization or re-establishment of viability (Tyler-Walters et al., 2001). Impacts are not quantified; rather spatially explicit maps of fishing effort per habitat type are compared to the habitat-specific recovery time to obtain an indication of the seabed area that is in a persistently disturbed condition (Ziegler and Valentinsson, 2008).

Foden et al. (2010) quantified the impact of three types of bottom-trawling on the UK seabed. Satellite-based vessel monitoring data were used to map the spatial-temporal distribution of trawling effort per gear-type (beam trawls, otter trawls, and dredges) per year ($\text{km}^2 \text{trawled year}^{-1}$). They used habitat specific and gear-type specific recovery times as a quantitative proxy for habitat sensitivity. These recovery rates were obtained from a literature review of empirical studies determining habitat- and gear-type specific recovery rates after cessation of trawling. Impacts are expressed as an estimate of the proportion of fished habitats in which recovery would be possible (i.e. full recovery before the next fishing disturbance) at the levels of fishing effort in a given year).

Veltman et al. (2011) quantified impacts of drill cutting disposal on benthic communities on the Norwegian continental shelf. Benthic impacts were quantified based on the initial benthic response and a recovery rate derived from empirical studies on recovery of benthic communities after particle disposal. The initial benthic response was quantified based on an SSD developed from laboratory studies on effects of particle deposition (burial) on several benthic species (Smit et al., 2008). Characterization factors are expressed as potentially affected fraction of species per volume of sediment.

Stelzenmüller et al. (2010) and Stelzenmüller et al. (2009) quantified impacts of aggregate extraction on the UK continental shelf based on mapping of pressure occurrence and a spatially-explicit average species sensitivity index. This index links species occurrence probability with an index for single species sensitivity, which is based on habitat characteristics (e.g. recoverability) and species characteristics (e.g. reproductive strategy, mobility). Potential impacts on the seabed are expressed as the total species sensitivity index per area.

A key advantage of the developed approaches is that they are based on a common framework, i.e. impacts of various physical pressures are quantified by coupling spatially-explicit maps of pressure occurrence with a (habitat-specific) benthic recovery rate as a proxy for ecosystem sensitivity. This approach can be consistently applied to a range of physical seabed pressures (e.g. bottom trawling, aggregate extraction, and particle deposition) and extrapolated to other (temperate) geographic locations. This allows for a meaningful, quantitative comparison of different types of activities which result in physical disturbance of the seabed and can be used to assess cumulative impacts, as shown by Foden et al. (2010). This provides an opportunity to develop LCA impact indicators for seabed damage and use at a mid-point level. This midpoint indicator would express the time-integrated loss of degraded habitat in $\text{km}^2_{\text{degraded habitat}} \text{year}^{-1}$. The requirement of high-resolution maps of both habitat type and pressure occurrence currently limits the applicability to western (European) regions as such data is presently not available at a global scale (Halpern et al., 2008; Christensen et al., 2009). The current application potential of these methods for LCA purposes is limited to temperate ecosystems (e.g. northern Europe). At present, taxonomic coverage is limited (Veltman et al., 2011; Stelzenmüller et al., 2010; Stelzenmüller et al., 2009) or not included (i.e. habitat recoverability only) (Foden et al., 2010; Nilsson and Ziegler, 2007).

Veltman et al. (2011) provide a method for the quantification of biodiversity loss arising from the transformation of seabed following particle deposition. The volume- and time-integrated impact of particle deposition on the seabed is calculated based on the initial benthic response (expressed in potentially affected fraction of species), a constant recovery rate and an assumed benthic species occurrence in the top 5 cm of the sediment (Dauwe et al., 1998; Flach et al., 2002). Such an approach could also be followed for other physical pressures. For example, the meta-analysis of Collie et al. (2000) on the vulnerability of benthic fauna to bottom trawling provides an opportunity to derive species sensitivity distributions (SSD) of the initial response of benthic communities to trawling.

This approach is analogous to the approach taken for quantifying land use transformation impacts and provides end-point indicators expressed in $\text{PAF m}^3 \text{ day unit intervention}^{-1}$. However, challenges arise from the inclusion of recovery within impact assessment methods. Recovery time depends on the attributes used (e.g. abundance, richness, diversity or evenness) and the definition of recovery (Botter-Carvalho et al., 2011). Complete recovery is the return of an ecosystem to its original, pre-disturbance state, whereby the abundance, diversity, structure and functioning of the biological community are the same as prior to the disturbance. However, for heavily disturbed seabeds, such as frequently trawled areas, the established recovery time can represent “recovery” to a point in a constant disturbance cycle rather than return to its pristine, pre-disturbance state (Kaiser et al., 2002; Foden et al., 2010). Capturing all aspects of recovery within an indicator is unrealistic. At present, a range of definitions and impact indicators are therefore available to characterize recovery. A standardized definition of recovery, and attributes to be considered, would be beneficial for improving the comparability of results from different LCIA methods.

6. (Over)exploitation

6.1. Cause effect

Overexploitation, more specifically overfishing, currently ranks as the top-threat to biodiversity loss in global marine ecosystems (Millennium Ecosystem Assessment, 2005; Costello et al., 2010). Fishing is characterized by an intense selective removal in terms of targeted species (Crowder et al., 2008; Pauly et al., 2002). This intense selective removal has direct effects on the target population. Recent analyses have documented a stark, global decline in abundance of pelagic and demersal predators of 50–70% and 90%, respectively, as a result of fishing activities (Crowder et al., 2008). Fishing can also have indirect effects

on the entire food web, for example, multi-level trophic cascades when top-predators are functionally removed from the food web and mid-level predators are released from predator control (Crowder et al., 2008; Pauly et al., 2002; Casini et al., 2009; Heithaus et al., 2008). The extent and magnitude of trophic cascades varies across communities and is not easily predicted as predators influence their prey both by inflicting mortality and by inducing (energy-intensive) anti-predator behavior (Heithaus et al., 2008).

Next to being species-selective, fishing is highly size-selective. Fishing gears are often designed to remove large fish, i.e. slow-growing and late-maturing individuals, and allow smaller ones to escape (Heino and Godø, 2002; Sharpe and Hendry, 2009). This size-selective removal can result in phenotypic changes in life-history traits of fished populations, such as length-at-age and maturation timing (Sharpe and Hendry, 2009; Neuheimer and Taggart, 2010). Fished populations are often disproportionately represented by fish maturing at a younger age and a smaller size (Sharpe and Hendry, 2009; Neuheimer and Taggart, 2010; Olsen et al., 2004). The rate of this phenotypic change is shown to be positively correlated with exploitation intensity and there is increasing evidence that these phenotypic changes can sometimes have a genetic basis (Sharpe and Hendry, 2009). Fishing also has direct effects on non-targeted species, including seabirds, sea turtles, marine mammals, and non-targeted fish species through by-catch (incidentally caught non-target species that are landed together with the target species) and discards (incidentally caught non-target species that are discarded) (Crowder et al., 2008; Heino and Godø, 2002).

6.2. Quantitative approaches and LCA perspective

Commercially exploited wild marine species fit within two LCA areas of protection (AoP): ecosystem quality and natural resources. LCA-specific assessment methods for the quantification of overexploitation impacts on marine fisheries have been proposed by Langlois et al. (2014) and Emanuelsson et al. (2014). The proposed methods have greater focus on addressing the natural resource AoP than the ecosystem quality AoP, although there is some overlap.

Langlois et al. (2014) proposed two methods for quantifying the impact of stock exploitation, one at the species-scale and the second at the ecosystem-scale. The species-scale assessment is based around the concept of stock management using a maximum sustainable yield approach (MSY). Depending on the status of the stock i.e. underexploited, fully exploited, over exploited, depleted or recovering, one of two characterization approaches are appropriate. For stocks that are under- or fully exploited the uptake of a mass of a given marine species (i.e. fish catch) is multiplied by the characterization factor $1/\text{MSY}$. Given that MSY is the maximum regeneration capacity of a fish stock in a given year, impacts are expressed as the maximum potential regeneration time (years). For example, if uptake of mass was equal to MSY the maximum potential regeneration time would be quantified as 1 year. For stocks that are overexploited, depleted or recovering from depletion the capture of a given amount of stock would have a larger impact than the capture of the same mass of a stock fished sustainably with the same MSY. As such, an additional factor (the ratio MSY/C_t , where C_t represents mean fish catches during the five years prior to impact assessment) is included that reflects the difference between MSY and current fish catch. Impacts are expressed as the maximum potential regeneration time (years). The characterization methods proposed by Langlois et al. (2014) require knowledge of stock status, MSY and, for overexploited, depleted or recovering stocks, mean fish catches during the five years prior to impact assessment. Stock assessment data are available in the RAM Legacy Stock Assessment Database (named in honor of Ransom A. Myers, a pioneering contributor to this database), which includes MSY values for 138 stocks worldwide (Ricard et al., 2012). The species-scale approach developed by Langlois et al. (2014) is applicable for the removal of fish from stocks managed using MSY. A method for accounting for non-targeted removal of marine mammals

could be developed based on the potential biological removal (PBR) approach. The PBR approach estimates the number of individuals of a species considered safely removable from the population without causing a reduction in population size below the minimum required for the conservation of the species (Lonergan, 2011).

The ecosystem-scale approach proposed by Langlois et al. (2014) involves the estimation of primary productivity embodied in the harvested biomass relative to the total amount of biomass in the ecosystem (area based estimation) and the rate of primary production in the cell from which biomass was harvested. Impacts are expressed in terms of time (years) required to regenerate the amount of biomass removed from the sea at the ecosystem level.

Emanuelsson et al. (2014) proposed lost potential yield (LPY) as a midpoint impact indicator for overexploitation. LPY reflects the difference between current and target fisheries management, indicates the impact on biotic resource availability and is a proxy for ecosystem impacts within each stock. LPY represents average lost catch owing to ongoing overfishing, assessed by simplified biomass projections covering different fishing mortality projections. Two simpler alternative methods were proposed that assess overfishing through 'fishing mortality' and 'overfishedness of biomass' respectively. LPY and the two alternative methods are primarily indicators of exploitation impacts on biotic resource availability and not 'ecosystem quality' per se. Emanuelsson et al. (2014) suggest that while the proposed LPY impact category is based on an anthropocentric resource-based perspective, it correlates with the ecosystem damage of extracting part of a stock. Therefore, LPY is comparable between stocks in terms of resource loss but only within stocks (i.e. temporal comparison) in terms of ecosystem damage (Emanuelsson et al., 2014).

Methods to address indirect effects of biomass removal on the foodweb, such as trophic cascades have not yet been developed in an LCA context. Employing an ecosystem model, such as Ecopath with Ecosim (Christensen and Walters, 2004), may present an opportunity for quantifying ecosystem-scale impacts. EcoOcean is a marine ecosystem model constructed using Ecopath with Ecosim. EcoOcean employs mass- and energy-balances to predict region-specific changes in abundance of 39 functional groups as a function of fishing mortality (Alder et al., 2007). These functional groups are common to the world's ocean and comprise multiple groups of primary producers, invertebrates, fish species, and marine mammals. EcoOcean can be employed to estimate changes in landings, stock depletion and a species depletion index. Biodiversity impacts of global fishing are expressed using the depletion index (DI), which provides a proxy for changes in species composition and mean abundance of original species belonging to an ecosystem (Alder et al., 2007). The DI evaluates changes in functional group biomass (estimated by the EcoOcean model) and potential species depletion based on the intrinsic vulnerability of species to fishing within the functional group: i.e. if EcoOcean predicts a decline in biomass of a functional group, it is assumed that species with greater vulnerability to fishing will experience larger depletion. The intrinsic vulnerability to fishing is available for 733 fish species in the *Sea Around Us* Project database. Polar regions are presently excluded from the modeling approach due to a lack of biomass and fishing effort data and incompleteness of catch data (Christensen et al., 2009; Alder et al., 2007). The model is thought to be less appropriate to address more specific, local-scale biodiversity changes due to the poor species resolution and the lack of local effort data (Christensen et al., 2009).

EcoOcean and the DI provide an interesting opportunity to develop LCA impact indicators that are highly compatible with PDF based impact indicators. However, species resolution is poor in EcoOcean, which prevents addressing specific biodiversity questions (Christensen et al., 2009). This could hamper its use in LCA when the aim is to compare fishery-derived products in detail. It is thought that the EcoOcean sub-models provide a well-defined starting point, but they should be enriched with local data (Christensen et al., 2009). Generic methods to quantify impacts of fishing on the target population, i.e. beyond

depletion of fish stocks, such as reduced size-at-age of mature fish, direct effects on non-target species and indirect effects on the foodweb, such as trophic cascades, are not yet available.

7. Invasive species

7.1. Cause-effect

An invasive species is a species occurring outside its natural range as a result of anthropogenic influence, i.e. introduced, that damages ecosystem quality. Invasive species may also damage the economy and/or human health (Lodge et al., 2006). The majority of introductions in the marine environment occur as a result of international shipping, principally via hull fouling and/or ballast water (stowaway pathways), aquaculture (bio-contaminant and escape pathways) and canal construction and operation, which facilitates dispersal across a geographic barrier (corridor pathway) (Gollasch, 2006; Molnar et al., 2008; Hewitt et al., 2009; Hulme et al., 2008).

The process of invasion occurs through several stages, i.e. introduction, establishment and spread. At each stage of invasion there are barriers that need to be overcome by a species or population to progress to the next stage (Blackburn et al., 2011; Kolar and Lodge, 2001). A species can fail or succeed at each barrier, with most invasions failing at one of the stages (Blackburn et al., 2011; Briski et al., 2011). For example, abiotic conditions of the recipient environment, particularly salinity and temperature, are a well-known barrier to the establishment of introduced species, limiting species to conditions approximating their native ranges (Olyarnik et al., 2009). The probability that a species successfully completes a transition depends on several interacting factors, including species traits, community invasibility, and factors unique to the specific introduction event, such as propagule pressure (the number and rate of individuals introduced) (Blackburn et al., 2011; Kolar and Lodge, 2001). Species abiotic niche requirements and propagule pressure have been consistently related to establishment success of invasive species across taxa and locations (Colautti et al., 2006; Lockwood et al., 2009; Johnston et al., 2009). Similar indicators of community invasibility that are comparable across taxa, locations and ecosystems, are not yet available (Catford et al., 2012). Case studies on human-mediated invasions have identified several factors influencing community invasibility, including resource availability (space and food), original community diversity, and degree of disturbance (Olyarnik et al., 2009). However, isolating the effect of the various factors of invasion success is difficult due to the confounding effects of high propagule pressure, which has hampered development of robust generalizations (Olyarnik et al., 2009; Sorte et al., 2010).

Documentation of the ecological consequences of invasive species is relatively scarce and comes mainly from studies conducted in the Eastern Mediterranean, the Black Sea and the Wadden Sea (Briggs, 2010; Finenko et al., 2006). In contrast to terrestrial and freshwater ecosystems, there are presently no known examples of local extinctions in marine ecosystems directly linked to an invasive species (Craig, 2010). However, invasive species do alter marine ecosystems to a large extent and have been shown to introduce diseases, displace native species, alter food-web dynamics, and drive local declines and shifts in habitat type (Briggs, 2010; Pyšek and Richardson, 2010; Crain et al., 2009).

7.2. Quantitative approaches and LCA perspective

Keller et al. (2011) coupled environmental niche- and vector-based models to assess "invasion risk" of global ports due to global shipping traffic. They used global databases of ship traffic and port environmental conditions to assess the chance that individual arriving ships will contain organisms capable of surviving in the recipient port. This approach is based on the known positive relationships of establishment success with i) the number of organisms released (propagule pressure), and ii) the environmental similarity between source and recipient

ecosystem (Keller et al., 2011). Two indicators of “invasion risk” are given, i.e. the ecological similarity index and the number of ships arriving at each individual port, which is used as a proxy for propagule pressure. The environmental similarity index is not easily linked to potential life cycle inventory data, for example kg goods transported by sea, in a meaningful manner. Assuming fixed trading routes, it would be possible to link a functional unit of transported goods to the number of ‘average ships’ arriving at each port, enabling development of a midpoint characterization factor: No. of ‘average ships’ $\text{kg}_{\text{good}}^{-1}$ transported by sea.

Hanafiah et al. (2013) proposed a framework for assessment of impacts, in an LCA context, resulting from invasive species introduction from inland shipping in a freshwater system. Characterization factors, expressed as potentially disappeared fraction (PDF) of native freshwater fish species in the rivers Rhine and Danube integrated over time per amount of goods transported ($\text{PDF m}^3 \text{ year kg}^{-1}$), were derived for exotic fish species introduction related to the transport of goods across the Rhine–Main–Danube canal. The fate factor expressed the time integrated change in fraction of exotic species due to a change in the transportation of goods and was approximated with empirical data as the change in exotic species occurrence (specifically caused by transport related activities) relative to the total species pool in a specific period, and the yearly average amount of transported goods between waterways in that time period. To calculate an effect factor, Hanafiah et al. (2013) constructed an empirical stressor–response relationship between the fraction of exotic species introduced and the fraction of native species threatened (IUCN red list species). As such, the effect factor reflects the impact of introduced species on native freshwater species richness in $\text{PDF m}^3 \text{ exotic species}^{-1}$. Data limitations, such as the rate of change in fraction of invasive (or exotic) species as a total of species within an ecosystem unit and the subsequent response of native species, e.g. in terms of fraction of threatened species, presently restrict the upscaling of this method to assess invasive species impacts of shipping in the marine environment.

8. Marine plastic debris

8.1. Cause–effect

World plastic production increased by almost 300% between 1989 (99 million tonnes) and 2012 (288 million tonnes) (PlasticsEurope, 2013). An estimated 275 million tonnes of plastic waste was generated in 192 coastal countries in 2010 (Jambeck et al., 2015). Mismanaged plastic waste has potential to be transported to, or released directly into, the marine environment (Jambeck et al., 2015). Plastics in the marine environment are of increasing concern because of their persistence and potential effects on marine biodiversity (Thompson et al., 2009).

Floating marine plastic debris is transported by prevailing winds and surface currents. Five accumulation sites, or gyres, at subtropical latitudes and related to overlying anti-cyclonic wind systems have been identified (Lebreton et al., 2012). Ultra violet radiation, waves and physical abrasion fragment marine plastic debris (Andrady, 2011), producing tiny fibers and granules, collectively termed microplastics. The upper size limit of microplastic varies between 1 mm and 10 mm (Cole et al., 2011). Fragments larger than microplastic are termed macroplastic.

An estimate of floating plastic abundance and weight, allocated by fragment size-class, across all five subtropical gyres and coastal Australia, Bay of Bengal and the Mediterranean Sea indicated that “the ultimate fate of buoyant microplastics is not at the ocean surface” (Eriksen et al., 2014). The deep sea, in particular seabed sediments, has been identified as a major sink for microplastic debris (Woodall et al., 2014). One mechanism through which microplastics sink is with phytoplankton aggregates (Long et al., 2015). Average microplastic sinking rates were found to increase from tenths of meters per day to hundreds of meters per day when aggregated with phytoplankton (Long et al., 2015).

Marine plastic debris interacts with organisms, including invertebrates, fish, seabirds and mammals, through multiple mechanisms; the most commonly reported of which are ingestion and entanglement (Gall and Thompson, 2015). Ingestion of plastic debris may reduce organism fitness by limiting food consumption (Ryan et al., 1988) for example by obstructing the gut or creating a false sense of satiation. Impacts associated with entanglement include injury, suffocation and general debilitation, which may result in decreased food consumption and reduced ability to avoid predators (Gregory, 2009). Other impacts attributed to marine plastic debris include smothering, i.e. sunken macroplastics inhibiting gas exchange and thereby creating patches of anoxic or hypoxic conditions on the seabed, transport of biota on floating debris, which could facilitate species introductions (Gregory, 2009), and facilitating transfer of toxins from seawater to organisms via ingestion (Teuten et al., 2009).

Research to date has focused on reporting of encounters between individual organisms and plastic debris. Evaluations scaling-up individual mortality to potential negative impacts on marine biodiversity at the population, community or ecosystem scale are lacking (Gall and Thompson, 2015).

8.2. Quantitative approaches and LCA perspective

Quantitative approaches that i) estimate the transport of land-based plastic waste to the marine environment (Jambeck et al., 2015), and ii) model the transport and accumulation patterns of plastic debris in the marine environment (Lebreton et al., 2012) have been developed. Jambeck et al. (2015) estimated the total quantity of land-based plastic waste entering the marine environment in 2010 as between 4.8 and 12.7 million tonnes. The estimate included waste created in 192 coastal countries and used a series of fractional factors. Fractional factors enabled estimation of the fraction of land-based waste that is plastic, mismanaged and transported to the marine environment. There is a paucity of information regarding the transport of mismanaged plastic waste to the marine environment. Jambeck et al. (2015) assumed that 15% to 40% of mismanaged plastic waste is transported to the marine environment. Lebreton et al. (2012) applied a global ocean circulation model coupled to a Lagrangian particle tracking model to simulate the transport, distribution and accumulation of floating debris in the marine environment from both land- and sea-based inputs.

Based on the work of Jambeck et al. (2015) and Lebreton et al. (2012), a fate factor could express the fraction of land-based plastic waste that is transported to the marine environment and to which accumulation zones they are transported according to the country of origin of the plastic waste. Such a fate factor would assume that all plastic in the marine environment was floating. In addition, given that mismanaged plastic waste is present in terrestrial and freshwater, as well as marine environments, it would be helpful to have a unified fate model covering all three receptors of plastic waste (Hoellein et al., 2014).

Currently there are no effect factors developed that quantify the effect of marine debris on biodiversity. Further research is therefore required to understand the sensitivity of marine species to the effects of plastic debris at the population community and ecosystem scales i.e. elucidating the significance of reported individual mortalities at larger scales. Effect factors would need to be differentiated according to fragment type due to their differing modes of impact. For example, the majority of entanglement encounters occur with macroplastic, such as rope and netting, whereas the majority of ingestion incidents occur with microplastic fragments (Gall and Thompson, 2015). In addition, time horizon considerations need to be improved through understanding of the degradation rate of plastics, and associated plastic fragment size-class distribution, and identification of plastic sinks, i.e. mechanisms through which the exposure of marine biodiversity to plastic debris are arrested.

9. Research outlook

Our review shows that model approaches to predict the spatial and temporal distribution and intensity of human interventions in the marine environment are relatively well-established. These models can be employed to develop LCA fate and exposure factors, spatially-explicit where appropriate, for each of the seven predominant stressors acting on that marine environment. Future research should focus on refinement of these fate models, particularly with respect to time horizon considerations and GCM and ocean-carbon cycle models. Specific research challenges are:

- To include the non-linear behavior of GCMs within an LCA framework
- To incorporate the time lag of climate change-induced ocean warming due to thermal inertia of the oceans
- To improve spatial differentiation, particularly the representation of coastal regions in GCMs and ocean-carbon cycle models and spatial variability of nutrient limitation in eutrophication models
- To generate data, such as the rate of change in fraction of species that are invasive within an ecosystem unit in response to anthropogenic activities, required to facilitate invasive species fate modeling
- To further advance our knowledge of the fate of plastic debris in the marine environment, in terms of degradation and loss rates, to determine the potential exposure duration.

Models to derive ecological response relationships are less developed (Table 1). For ocean acidification, eutrophication-induced hypoxia and particle deposition on the seabed, however, species sensitivity distributions (SSDs) can be employed, based on available effect data from lab tests and/or field observations. This allows development of LCA effect factors in terms of potentially affected fraction of species. Indicators relating to loss of species richness, analogous to potentially disappeared fraction of species, could theoretically be produced for climate change (on a global scale), overexploitation (on a regional scale) and invasive species (on a local scale). Other indicators of impacts on biodiversity resulting from climate change include species range shifts (on a global scale) and habitat loss (local scale, specific to warm water coral reefs). Currently available indicators of the biodiversity impact of overexploitation are currently limited to illustrating the level of perturbation that biomass removal has on individual stocks of commercial species or provide a general indication of a reduction in biodiversity support function of the ecosystem: i.e. loss of energy available to the whole ecosystem. Neither of these indicators of overexploitation provide a robust indication of effects on biodiversity. Indicators for effects of marine plastic debris on biodiversity are currently lacking completely. In order to improve existing effect modeling approaches we recommend:

- Incorporating a larger number of taxonomic groups, particular marine mammals and those from polar regions, within the GCM-biotic niche modeling approach
- Further investigation to better understand the mechanistic cause-effect pathway from ocean carbonate chemistry to organism fitness and survival. Such investigation is especially required for non-calcifying organisms
- Establishment of species-response relationships to exposure to plastic debris, invasive species and removal of biomass
- Standardization of the definition of ecosystem recovery following stress of finite duration to facilitate estimating recovery rates and the recovered state.

Approaches for quantifying the effect of stressors on marine biodiversity include: PAF, PDF, area habitat loss, species range shifts, loss of energy for ecosystem functioning (biodiversity support) and measures of perturbation to single-species commercial stocks; at local, regional or global scales. There is a need to identify how impact indicators for

drivers of marine biodiversity loss could be standardized in order to facilitate comparison and/or integration across stressors, geographic locations, ecosystems and spatio-temporal scales.

The current limited coverage of stressors on marine environments within LCA results in an under-representation of actual impacts on marine environments. Where stressors are not covered, impacts are often assumed negligible. Whilst the quantitative approaches and novel LCIA techniques discussed in previous sections are associated with limitations and high levels of uncertainty, they provide a foundation from which impacts on the marine environment can be assessed within the LCA framework. We call upon the research community to work towards better representations of impacts on marine biodiversity in LCA in order to close this important gap.

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