

# Naturalness as a basis for incorporating marine biodiversity into life cycle assessment of seafood

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## Abstract

**Purpose** Methods to quantify biodiversity impacts through life cycle assessment (LCA) are evolving for both land- and marine-based production systems, although typically independently from each other. An indicator for terrestrial food production systems that may be suitable to assess marine biodiversity, and is applicable across all food production systems, is a measure of hemeroby or distance from the natural state. We explore the possibility of adapting this approach to marine systems to assess the impact of fishing on seawater column and seafloor systems.

**Methods** The terrestrial hemeroby concept is adapted here for marine ecosystems. Two commercial fishery case studies are used to trial the effectiveness of hemeroby in measuring the influence exerted by fishing practices on marine biodiversity. Available inventory data are used to score areas to a hemeroby class, following a semi-quantitative scoring matrix and a seven-point scale, to determine how far the seafloor and seawater column are from their natural state. Assessment can progress to the impact assessment stage involving characteri-

sation of the hemeroby score, to determine the Naturalness Degradation Potential (NDP) for use in calculating the Naturalness Degradation Indicator (NDI). The method builds on well-established processes for assessing fisheries within the ecosystem-based fisheries management framework and is designed to enhance assessment of fishing impacts within LCA.

**Results and discussion** Australian fisheries case studies were used to demonstrate the application of this method. The naturalness of these fisheries was scored to a hemeroby level using the scoring matrix. The seafloor of the Northern Prawn Fishery and the seawater column of the South Australian Sardine Fishery were both classified as partially close to nature. Impact assessment was carried out following the process outlined for the NDI. The naturalness degradation results were highly sensitive to area calculation method and data. There was also variation in results when using annual or averaged data for catch. Results should therefore be interpreted in the context of these sensitivities.

**Conclusions** Adaptation of the hemeroby concept to marine habitats may present an opportunity for more informed comparison of impacts between terrestrial and marine systems. Incorporating a measure of naturalness into assessments of food production can be useful to better understand the cost, in terms of transforming ecosystems from natural to more artificial, of meeting growing food demand. Biodiversity is a broad concept not easily captured through one indicator, and this method can complement emerging biotic LCA indicators, to provide a suite of indicators capable of capturing the full impact of fishing on marine biodiversity.

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## 1 Introduction

Terrestrial and aquatic habitat change have been identified as direct drivers of biodiversity loss, primarily as a result of land use for agriculture and marine use for fishing (Millennium Ecosystem Assessment 2005). Land use is a priority impact category in life cycle assessment (LCA) (Jolliet et al. 2014), and several methods for the quantification of land use in LCA are in advanced stages of development, although a consensus on best practice is yet to be reached (Teixeira et al. 2016; Michelsen and Lindner 2015). Methods to quantify comparable impacts on biodiversity in aquatic habitats have not yet been formalised (Curran et al. 2010). The UNEP/SETAC *Framework for Land Use Impact Assessment within LCA* proposes that physical changes in the seabed be considered as land use-related impacts for marine systems, while the biological effects of fishing be considered under the depletion of biotic resources (Milà i Canals et al. 2007). Fishing directly affects both pelagic (water column) and benthic (seafloor) ecosystems (Halpern et al. 2008), and while land is considered an essential support of terrestrial ecosystem services, some life support functions in marine ecosystems are not directly related to the seafloor (Charpy-Roubaud and Sournia 1990). Many marine species have facultative, rather than essential, habitat associations where the seafloor is used for many important life processes, but the absence of these habitats does not result in species extinctions (Foley et al. 2012). Assessments of marine biodiversity therefore need to include aspects of quality and functioning of the whole water column, in addition to biodiversity structures such as sea floor habitat (Deros et al. 2007).

### 1.1 Methods for assessing impacts on biodiversity within LCA

Land use and conversion leading to loss in species richness is commonly modelled as an endpoint category in LCA to assess impacts on biodiversity (Souza et al. 2015). Biodiversity impact can be measured by counting species; however, the full impact is not always captured using this approach (Coelho and Michelsen 2014; Langlois et al. 2011; Millennium Ecosystem Assessment 2005) and habitat configuration and use intensity-based considerations are omitted (Teixeira et al. 2016) as well as functional ecosystem roles (de Souza et al. 2013). Alternative methods to assess impacts on biodiversity within LCA have examined relative areas used within ecosystems (Michelsen 2008; Curran et al. 2010) and species-area relationships (Chaudhary et al. 2015). Other authors have focussed on life support functions by measuring the biological production capacity (Langlois et al. 2011; Langlois et al. 2016; Hélias et al. 2014) or on the quality of land using a measure of naturalness (Brentrup et al. 2002; Fehrenbach et al. 2015).

A number of quantitative and qualitative methods have been proposed to assess the impacts of fishing on different aspects of biodiversity within the LCA framework. Ziegler et al. (2003) estimated the area of seafloor swept per kilogram cod using trawl dimensions, average boat speeds and reported fishing effort in the Baltic Sea. Ellingsen and Aanonsen (2006) also calculated total area swept for cod and compared results with land area required to produce grain for chicken and salmon feeds. Results were presented in square metres and did not include damage assessment, thereby implying that the impact of trawling on an area of seafloor was directly comparable to the impact from converting natural terrestrial habitat to monoculture grain production. A measure of total area swept can be more meaningful when fishing data are overlaid on habitat maps to calculate estimates of area affected by fishing that reflect resilience of the habitat and when aggregation of fishing effort is considered (Nilsson and Ziegler 2007). Several studies have since incorporated this method; however, results continue to be presented as area of seafloor impacted and not weighted according to intensity or resilience (Ziegler and Valentinsson 2008; Vázquez-Rowe et al. 2012a; Ziegler et al. 2011; Ziegler et al. 2015).

A Life Cycle Impact Assessment (LCIA) method to measure the environmental resource footprint of marine area occupation has been proposed for natural marine systems, where the ‘exergy’ content (the maximum work a system can deliver in equilibrium with its environment) of extracted resources is quantified (Taelman et al. 2014). A ‘sea use’ impact category has also been developed to assess transformation and occupation impacts in marine ecosystems as a measure of the free Net Primary Production (fNPP) (Langlois et al. 2011; Langlois et al. 2014a, b; Langlois et al. 2016). Quantifying the amount of primary production required to produce seafood, and the pressure placed on ecosystems by overfishing, has been explored by several authors; however, current practice typically does not consider species- and ecosystem-specific factors (Cashion et al. 2016). This approach also faces the challenge that seafood production can rely on primary production from areas distant to the location of the harvest, such as where the production of bivalves relies on primary production of microalgae carried by currents. Aspects of biomass removal by fishing have been explored in LCA through the quantification of the biomass that would not be produced in the future due to current overexploitation (Emanuelsson et al. 2014), measurement of the amount of threatened fish discarded per landed kilo of seafood (Hornborg et al. 2013) and estimates of bycatch and discards (Vázquez-Rowe et al. 2012b; Ziegler and Valentinsson 2008; Ziegler et al. 2011). These approaches to biomass removal also have complexity in their application, for example when fisheries occur with no cost to future production yet still alter the ecosystem from its natural state, for example when a target stock is depleted by fishing but is able to rapidly recover when fishing stops.

## 1.2 Fishing impacts on seafloor and seawater column biodiversity

Knowledge of marine biodiversity is increasing; however, large gaps remain (Costello et al. 2010). There is evidence that the form and function of marine ecosystems can be sustained at a wide variety of fishing pressures, including some cases with very high depletion of the target species; however, the structure of ecosystems under extreme fishing pressure is usually highly modified (Hilborn et al. 2015). Bottom trawl fisheries impact benthic communities via the dragging of fishing gear over the seafloor, and the FAO estimate that 23% of the global capture production is obtained from these fisheries (FAO 2009). Trawling activity has intensified and spread since the 1950s when global records were first assembled (Watson et al. 2006). While bottom trawling has been likened to clear felling of forests (Watling and Norse 1998), the effects of trawling vary widely depending on the vulnerability and recovery rates of benthic species and structures (Collie et al. 2000; Kaiser et al. 2006; Althaus et al. 2009). Trawling can cause severe damage to some benthic habitats, such as seamounts (Williams et al. 2010). Yet, at the other extreme, it can be benign on habitats where the benthos is resistant to trawling, particularly in areas where trawl and natural disturbance affect benthic communities in similar ways (van Denderen et al. 2015).

Not all fishing directly impacts the seafloor. Many fisheries operate entirely within the pelagic zone using gear such as purse seine, pelagic trawl, surface gill nets and midwater longline, with only incidental impacts on the seafloor through, for example lost gear and anchoring. Although these fisheries barely interact with the seafloor, they nonetheless can alter the natural system through the removal of biomass. Low fishing pressure typically reduces the average abundance of species without altering ecosystem functions; however, higher exploitation rates can cause changes in trophic structure and very high rates may even lead to depensation and local extinctions (Hilborn et al. 2015). Fishing has also been linked to evolutionary changes in exploited fish stocks, a process which is not currently incorporated into management models (Zimmermann and Jørgensen 2015). Fish stocks do not exist in isolation, and, if poorly managed, fishing can place populations of both target and non-target species at risk (ICES 2005). Ecosystem functioning of the whole water column should therefore be considered in assessments of naturalness of the marine environment (Derous et al. 2007) and has been recognised through ecosystem-based fisheries management (Pikitch et al. 2004) and in the assessment approach of independent sustainable seafood certification bodies, such as the Marine Stewardship Council (MSC) (Lack 2004; Mayfield et al. 2014).

## 1.3 ‘Naturalness’ and the hemeroby concept

Natural ecosystems are identified as one of the safeguard subjects, or areas for protection, in LCA (Consoli 1993), and several authors have proposed indicators that incorporate a measure of naturalness of agricultural and forestry production systems within the LCA framework (de Souza et al. 2013; de Baan et al. 2013; Rüdiger et al. 2012; Michelsen 2008). The term *hemeroby* is used in landscape ecology to express distance to nature and has been identified as a consistent method for use in LCA that captures the complexity of land use, with an acceptable level of simplification and without loss of crucial information (Fehrenbach et al. 2015). The implementation of hemeroby, and other indices that simplify complex changes, however, is associated with a trade-off in the level of ecosystem complexity in order to make generalisations to classify landscapes. The hemeroby concept provides a measure of naturalness in a system where the lowest values (ahemerob) correspond to ‘natural’ or non-disturbed landscapes and the highest values (metahemerob) are given to totally disturbed or ‘artificial’ landscapes. The hemeroby concept has been adapted for use in LCA to account for the decreasing availability of habitats (Brentrup et al. 2002), to capture the complexity of land use (Fehrenbach et al. 2015) and in combination with other land use indicators (Coelho and Michelsen 2014; Taelman et al. 2016). Naturalness is a complex, multidimensional concept, and combining qualitative information on ecosystem quality with quantitative approaches can create a more comprehensive picture (Hochschorner and Finnveden 2003). Several indices have been proposed to express how impacts from agriculture and forestry move the state of the landscape away from the natural one, therefore allowing for characterisation of different types of land use (Machado 2004; Fehrenbach et al. 2015; Brentrup et al. 2002). No such index currently exists to express how impacts from marine production move the state of the ecosystem away from the natural one, although an index for water bodies is reportedly in the early stages of development (Fehrenbach et al. 2015). Adaptation of the hemeroby concept to marine habitats may present an opportunity to overcome the lack of consideration for marine systems in current LCIA methods (Taelman et al. 2014) and provide a more informed comparison of impacts between terrestrial and marine production systems.

## 1.4 Adapting the hemeroby concept to marine habitats

The concept of naturalness can be applied either as a conservation value or as a parameter, or state descriptor, of ecosystems (Machado 2004). The latter is adopted in this research, and it is assumed that the closer to ahemerob (natural) a

production system, the better, in relation to ecosystem functionality and biodiversity, although in terms of food production systems, the goal is not to achieve a natural state, as may be the case with a conservation objective. The potential use of the hemeroby concept is examined here as a proxy for the influence exerted by fishing practices on marine biodiversity and the potential application of a Naturalness Degradation Indicator (NDI) using two commercial fishery case studies. A challenge of this research is that marine ecosystems are subject to regime shifts at different time scales (Rocha et al. 2015). This variation through time can complicate identification of the ahemerob (natural) state and original biodiversity. Similar challenges arise in terrestrial ecosystems (Folke et al. 2004) from where the hemeroby method was derived.

We propose the application of a NDI to semi-quantitatively score the impact of the fisheries on the seafloor and seawater column. Incorporating a measure of marine naturalness into LCA can complement recently developed seafood indicators, such as those addressing impacts from fishing on biomass production capability, using the production target of maximum sustainable yield (MSY), the point at which the highest fish catch can be sustained in the long term, as a reference point (Emanuelsson et al. 2014; Langlois et al. 2014a). Combining indicators that focus on ecosystems, habitats and fish stocks is also consistent with the key framework for land use impact assessment within LCA which recommended that the physical impacts of fishing should be assessed from both ‘natural environment’ and ‘resource’ perspectives (Milà i Canals et al. 2007).

The development of this NDI for marine systems is also intended to progress the field of seafood LCA research in a manner that is compatible with more general fishery management assessments. Much work has been done in the area of ecological risk assessment (ERA) to assess the broader ecological impacts of fishing. For example, the Ecological Risk Assessment for the Effects of Fishing framework has been applied to over 30 fisheries in Australia and elsewhere and has been adapted for use by the MSC, in particular for use in data-deficient fisheries (Hobday et al. 2011). Current independent seafood certification processes and the ERA process are based around meeting the anthropocentric fishery management objectives and the naturalness of the ecosystem, and habitat within which the fishery operates is not measured. Here, we combine the hemeroby approach with established peer-reviewed criteria developed by the MSC to create a semi-quantitative method for LCIA. The method demonstrates a different, but complementary, perspective which captures the naturalness of marine habitats and ecosystems using a measure of how far removed they are from an unfished state. In developing and demonstrating this method, we identified a number of challenges that require further work before this method can be fully operationalised.

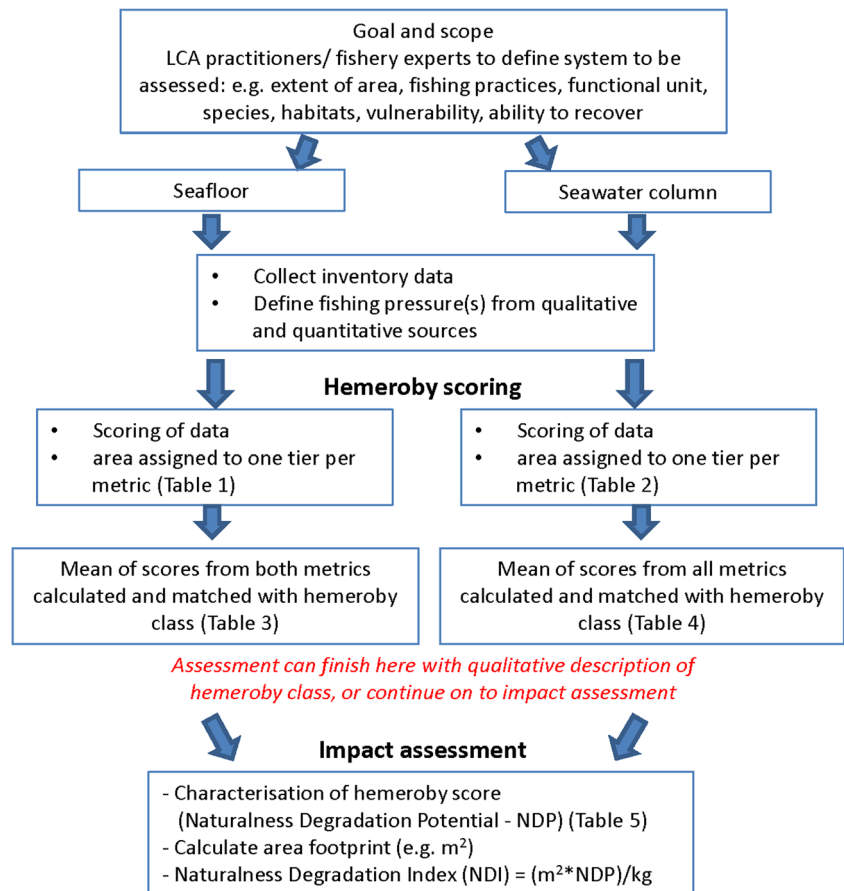
## 2 Methods

### 2.1 Index of naturalness

A Naturalness Degradation Index (NDI) is proposed here to classify impacts on marine biodiversity from fishing. The process underpinning application of the NDI, outlined in Fig. 1, begins with the goal and scope of the LCA which will determine whether or not the assessment will continue past the determination of hemeroby class to the impact assessment. The system to be assessed is then defined in terms of the area to be assessed, species and habitats, their vulnerability to fishing and recovery potential. Seafloor and seawater column areas are split at this stage, and the impacts from fishing are defined for each and recorded as inventory data. These inventory data and information available on the habitat and species will vary for individual fisheries but may be sourced from resources such as fishery status reports, ecological risk assessments and academic publications. In situations where data are limited, qualitative assessment and expert judgement, including analogy from similar situations, can be used. Available inventory data are then used to score areas to a hemeroby class, following a scoring matrix and a seven-point scale, to determine how far the seafloor or seawater column is from its natural state. Impacts on demersal fish that live on the seafloor are captured under the seawater column scale while the impacts to their habitat are captured under the scale for seafloor. For pelagic fisheries that operate in the seawater column only, the assessment can be conducted using the water column scale. For fisheries that interact with both the water column and the seafloor, both scoring scales can be used and the scores from both assessments are averaged to provide an overall result.

To create the marine hemeroby scale, published scales developed for terrestrial systems (Brentrup et al. 2002; Fehrenbach et al. 2015; Walz and Stein 2014; Steinhardt et al. 1999) were examined and modified based on fishing impacts in marine environments. A seven-point scale is commonly adopted (Walz and Stein 2014) although as the classification of distance to nature can be made at different levels, for example by habitat or land use classes, variations to hemeroby scales can be found in the literature. Determination of an area’s hemeroby class may be the final assessment step and would result in a qualitative description of the system. Depending on the purpose of the study, as defined in the LCA goal and scope, the assessment may progress to the impact assessment stage. This step involves characterisation of the hemeroby score, to determine the Naturalness Degradation Potential (NDP). The NDP is multiplied by the area fished and divided by the functional unit in the final stage of calculating the NDI.

**Fig. 1** Proposed seven-step process for allocating seafloor and seawater column areas to hemeroby classes and calculating naturalness degradation impact for marine areas



## 2.2 Fishery case studies

Two Australian fisheries were selected as case studies to demonstrate the application of this method. For a seafloor case study, the Northern Prawn Fishery, a Commonwealth trawl fishery located in northern Australia, was used. The fishery is currently certified as sustainable by the MSC and not considered to be overfished (Patterson et al. 2015). Otter trawl gear is used to target a range of tropical prawn species; however, white banana prawn (*Fenneropenaeus merguensis*) and two species of tiger prawns (*Penaeus esculentus*; *Penaeus semisulcatus*) account for around 80% of the landed catch (Patterson et al. 2015). The Northern Prawn Fishery is characterised by a number of benthic habitats including reef platforms, soft sediments and deep siliciclastic deposits. The fishery covers an area of approximately 700,000 km<sup>2</sup>; however, it is estimated that less than 10% of the total area is trawled (Zhou and Griffiths 2008). Prawn fishing grounds of the Northern Prawn Fishery tend to be located on soft sediments that are resilient to perturbation by trawl gear (Bustamante et al. 2010). Unconsolidated sediments are more resilient than other habitat types such as reef, sponges and seagrass; however, these habitats are variable with some

unconsolidated sediments less resilient than others (Kaiser et al. 2006). 2.1% of the total area is never trawled due to permanent area closures, including all shallow water seagrass. Areas that are unsuitable for trawling, such as large reef outcrops and areas with low density of the target prawn species, are also not trawled (AFMA 2013). The impacts of sparse and infrequent trawl effort in this fishery are not currently considered a threat to biodiversity (Pitcher et al. 2009).

For the seawater column case study, the Australian sardine (*Sardinops sagax*) purse seine fishery was used. Australian sardines are a schooling pelagic fish species found in temperate waters between Rockhampton in Queensland and Shark Bay in Western Australia, including northern Tasmania on the continental shelf. The waters off South Australia support the largest component of the Australian sardine population, and sardines have been taken from these waters for use as live bait in the southern bluefin tuna fishery since the 1960s (SASIA 2012). A dedicated purse seine fishery has been in operation since 1991 and is managed in accordance with a harvest strategy with a set total allowable catch (TAC) and established ecological target and limit reference points (PIRSA 2014). The South Australian sardine stock is currently fished within sustainable limits (Ward et al. 2015).

### 2.3 Seafloor and seawater column assessment

The scales presented in Tables 1, 2, 3 and 4 were adapted from the literature by the authors in consultation with an expert working group consisting of six scientists from the Institute for Marine and Antarctic Studies (IMAS) at the University of Tasmania and the Australian Commonwealth Scientific and Industrial Research Organisation (CSIRO) with expertise in fisheries ecology and biology, population modelling, and assessment of marine habitats, biodiversity and ecosystem function. For each hemeroby class on the scale, a description is provided for types of seafloor and water column use with an indicative example. The scales were designed to allow comparison with terrestrial land use (see Electronic supplementary material).

#### 2.3.1 Criteria and metrics

The classification system developed here to score fisheries and assign results to a hemeroby class is based on the MSC scoring framework. The MSC has developed a Fisheries Assessment Methodology (FAM) based on three principles: (1) maintaining the productivity of fish stocks; (2) maintaining the structure, productivity, function and diversity of the ecosystem on which the fishery depends; and (3) effective management that meets the requirements of laws and standards and operational frameworks that require responsible and sustainable use of fish stocks (MSC 2002). Each principle is broken down into criteria with specific performance indicators (PIs) to be met for MSC certification.

The seawater column scoring metrics used here were based on criterion from the MSC's first and second FAM principles (MSC 2010). The third principle is not used as it is based on anthropocentric targets, such as 'sustainability' which are not relevant to the assessment of the naturalness of the systems examined here. The 'target species'; 'retained species'; 'by-catch'; 'endangered, threatened, protected (ETP) species' and 'ecosystem' PIs from the two FAM principles were used here for metrics. The 'habitat' PI was not included in the seawater column assessment as this usually refers to bottom habitats not the water column (MSC 2010). One metric per criterion was developed each with seven tiers ranging from zero to six, where zero represents a close-to-nature score while six is a distant-to-nature score. The authors, in consultation with the scientific expert group, developed seven tiers ranging from zero to six (Table 1) and their descriptions.

Biomass reference points were established for ecosystem thresholds. A common method of specifying biomass reference points is to express them as a percentage of the unfished, virgin biomass (%B<sub>0</sub>). For target species, 20% B<sub>0</sub> is often cited as a default reference point for the minimum acceptable biological limit (Rosenberg 1996; Hilborn and Stokes 2010). However, it is also recognised that the adoption of 20% B<sub>0</sub> is

unlikely to be applicable across the entire range of observed levels of stock resilience (Hilborn and Stokes 2010). An upper limit of 75% biomass can be used to identify the point at which impacts on trophic structure and ecosystem stability would be small (Smith et al. 2011; Salcido-Guevara et al. 2012). These figures are used as stock thresholds because of their wide application in fishery assessments; however, their use in this context has a different objective, which is assessing naturalness rather than productivity.

Seafloor scoring metrics were based on criterion from the second FAM principle. The ecosystem and habitat PIs were used for metrics as these were the most relevant to the seafloor condition (MSC 2010). The benthic metrics are focussed on impacts from fishing on habitat and not on biotic impacts. The impacts to populations of bottom fish, therefore, would be considered under the biotic impacts as part of the water column assessment. One metric per criterion was developed with tiers applied to each metric to assign the measurement (Table 2), as for the seawater column.

As data or evidence of impacts are not always available, a precautionary approach is taken in that fishing activities are assumed to pose higher risks in the absence of information, consistent with MSC (MSC 2010) and Ecological Risk Assessment for the Effects of Fishing frameworks (Hobday et al. 2011). Levels of confidence around impacts are incorporated into the assessment to help discern between hemeroby levels. For example, a fishery would be assigned to tier 2 of the ecosystem criterion if there is a high degree of confidence that the fishery is unlikely to cause serious harm; however, if there was low confidence in this impact, the fishery would be assigned to tier 3. Conversely, where there is confidence that fishing adversely affects a criterion, the result is allocated to a higher hemeroby class than if there was no confidence around adverse impacts. For example, a fishery would be assigned to tier 4 in the previous example if there was evidence that the fishery causes serious harm and 3 if there was no evidence of serious harm. The time required for ecosystems to recover is based on their capacity to recover, that is, ecosystems that can recover quickly are deemed resilient and those that are slow to recover are less resilient. A clarification of terms used to assess confidence is provided in the Electronic supplementary material (Table S3) as well as definitions of other terms used for scoring.

#### 2.3.2 Scoring

The area being assessed is classified into one tier for each metric (e.g. one tier for habitat, one tier for ecosystem, etc.). The number of the tier represents the same number of points, i.e. classification into tier 1 is associated with one point, tier 2 with two points and so on following Fehrenbach (2015). The overall score across all metrics is reported as the arithmetic mean. The resulting score between zero and six (Table 1 for

**Table 1** Seawater column-related criteria and metrics—the results for the area under investigation is assigned to the respective tier for each criterion

Criteria	Metrics
Criterion 1: target species	Metric 1: target stock biomass 0. Unfished, 100% virgin biomass (B0) 1. 75–99% B0, high degree confidence 2. 30–74% B0, high degree confidence 3. 20–30% B0 4. Around 20% B0, low degree confidence 5. <20% B0, high degree confidence 6. Species extinct
Criterion 2: retained/non-target species	Metric 1: level and reversibility of harm to the retained species 0. No retained species 1. Evidence that the level of harm is well below established reference points and has never approached limits 2. Low level of harm, within the limit set by established reference points, high degree of confidence 3. Likely that species could be depleted or recovery hindered—no reference points, low confidence 4. Species seriously depleted and/or recovery hindered, outside established reference points 5. Populations are functionally extinct, no retained species as a result of harm to population 6. Population extinct
Criterion 3: bycatch and discard species	Metric 1: level and reversibility of harm to bycatch species or species groups 0. No bycatch 1. Evidence that the bycatch level is well below established reference points and has never approached limits 2. Low level of harm, within the limit set by established reference points, high degree of confidence 3. Likely that bycatch species could be depleted or recovery hindered—no reference points, low confidence 4. Bycatch species seriously depleted and/or recovery hindered, outside established reference points 5. Bycatch populations are functionally extinct, no bycatch as a result of harm to populations 6. Populations extinct
Criterion 4: endangered, threatened, protected (ETP) species	Metric 1: level and reversibility of harm to ETP species 0. No interaction with ETP species 1. Evidence that the risk to ETP species is well below established reference points and has never approached limits 2. Low level of risk, within the limit set by established reference points, high degree of confidence 3. Level of harm likely to impact protection and rebuilding—no reference points, low confidence 4. Level or harm in excess of requirements for protection and rebuilding, risk to ETP species would be high 5. ETP populations are functionally extinct, ETP species not present as a result of harm to populations 6. Populations extinct
Criterion 5: ecosystems	Metric 1: level and reversibility of harm to the key elements of ecosystem structure and function 0. Unfished 1. Evidence that fishery does not affect overall biodiversity (species, community composition and structure) 2. Fishery unlikely to cause serious harm, effects are reversible in the short term, high degree confidence 3. Likely that fishery causes harm, effects are reversible in the long term 4. Evidence that fishery causes serious harm, effects are unlikely to be reversible in the long term 5. No resemblance to original ecosystem 6. Ecosystem has been completely destroyed

seawater column or Table 2 for seafloor) is then matched with the corresponding hemeroby class (Tables 3 or 4). Cut-off points were set at 0.5, for example a score of or between 1.5 and 2.4 will be allocated to hemeroby class 2 (partially close to nature). Class 6 may not be reached through fishing but has been included in the index as an indication of activities at that level of pressure, following the inclusion of a comparable artificial level in scales developed for agriculture (Fehrenbach et al. 2015; Brentrup et al. 2002).

Hemeroby data can be expressed as ordinal classes, such as those presented in Tables 3 and 4 or as discrete numbers. Life cycle inventory data for naturalness are commonly reported classified according to the respective hemeroby class, although the aggregation into a single indicator value is useful for certain applications (Fehrenbach et al. 2015), such as comparisons between different systems, and has been proposed for use in LCIA by several authors (Fehrenbach et al. 2015;

Brentrup et al. 2002; Taelman et al. 2016). One potential application is an impact assessment of seafood. The process of developing characterisation factors to represent the NDP is demonstrated for use in a Naturalness Degradation Indicator (NDI) for marine biodiversity. The NDI calculation is a function of the life cycle inventory data for the area fished ( $m^2 \times 1 \text{ year}$ ) multiplied by the appropriate characterisation factor (NDP) (Table 5) and divided by the functional unit, in this case, kilograms of catch ( $[m^2 \times NDP]/kg$ ). We demonstrate how the NDI might be applied; however, we note that the method requires further research and actual results should be interpreted with caution.

## 2.4 Characterisation

In the LCA assessment standard ISO 14044, it is recommended that characterisation factors reflect ‘a distinct identifiable

**Table 2** Seafloor criteria and metrics—the results for the area under investigation is assigned to the respective tier for each metric

Criteria	Metrics
Criterion 1: habitats	<p>Metric 1: level and reversibility of harm to habitat structure</p> <ol style="list-style-type: none"> <li>0. No interaction with habitat structure</li> <li>1. Evidence that there is no harm to habitat structure from fishing</li> <li>2. Fishery unlikely to cause serious harm, effects are reversible in the short term, high degree confidence</li> <li>3. Fishery likely to cause serious harm, i.e. alteration of habitat cover/mosaic, effects are reversible in the long term</li> <li>4. Evidence fishery causes serious harm, effects are unlikely to be reversible in the long term, i.e. recovery may not automatically occur</li> <li>5. No resemblance to original habitat</li> <li>6. Habitat has been completely destroyed</li> </ol>
Criterion 2: ecosystems	<p>Metric 1: level and reversibility of harm to the key elements of ecosystem structure and function</p> <ol style="list-style-type: none"> <li>0. Unfished</li> <li>1. Evidence that fishery does not affect overall biodiversity (species, community composition and structure)</li> <li>2. Fishery unlikely to cause serious harm, effects are reversible in the short term, high degree confidence</li> <li>3. Likely that fishery causes harm, effects are reversible in the long term</li> <li>4. Evidence that fishery causes serious harm, effects are unlikely to be reversible in the long term</li> <li>5. No resemblance to original ecosystem</li> <li>6. Ecosystem has been completely destroyed</li> </ol>

environmental mechanism and/or reproducible empirical observation’ (ISO 2006). Fehrenbach et al. (2015) allocated factors to reflect the exponentially longer periods of time that natural habitats require to develop. Their approach was also based on current area mix determined through assessments of global land use (UNEP 2014). There is no compatible assessment for the seafloor or seawater column. A linear approach is therefore used, following Brentrup et al. (2002), where intervals between the classes were constant at 0.167.

## 2.5 Area fished

Three methods were used to calculate bottom trawl ‘footprint’ from trawl effort data for the Northern Prawn Fishery (Pitcher et al. 2016):

- Total area of trawl footprint calculated from data gridded at 0.01° within a specified depth range, covering the total area of cells where trawling was recorded in the past 5 years

**Table 3** Definition and description of hemeroby classes for the seawater column

Code	Class	Description and indicative example of seawater column use and condition
0	Natural	- Seawater column in pristine or near pristine condition - No fishing influence
1	Close to nature	- Seawater column in natural state, natural species composition - limited removal of species through very low-intensity fishing
2	Partially close to nature	- Seawater column routinely used for fishing - E.g. low- to moderate-intensity purse seine
3	Semi-natural	- Intensively used seawater column - E.g. moderate to high intensity midwater trawl
4	Partially distant to nature	- Permanently altered seawater column, natural ecosystem severely impacted, especially where there are vulnerable and slowly recovering species, or replaced by invasive or exotic species - E.g. destructive practices or overfishing
5	Distant to nature	- Seawater column ecosystem highly modified, no resemblance to original ecosystem, e.g. highly polluted, with no natural biota or communities - Fishing influence prevents regeneration
6	Non-natural artificial	- No remaining ecosystem structure or function, e.g. reclaimed land with no habitat for marine species - No relevant fishing influence

This table to be used in combination with Table 1. Full scale with land use descriptions in Table S1



**Table 4** Definition and description of hemeroby classes for the seafloor

Code	Class	Description and indicative example of seafloor use and condition
0	Natural	- Seafloor in pristine or near pristine condition subject to only minor indirect influence, e.g. marine debris. Habitats types include highly remote (e.g. oceanic atolls) and very deep habitats (lower continental slope, continental rise and abyss—depths below 2000 m) - No fishing influence
1	Close to nature	- Seafloor in a natural state and populated by natural species; negligible historical direct impact - Minor (localised or short term) fishing influence, e.g. anchoring, single line methods
2	Partially close to nature	- Seafloor used routinely for human uses - E.g. low to moderate intensity activities, e.g. demersal trawl in resilient habitat, Danish seine in unconsolidated sediments
3	Semi-natural	- Intensively used habitats, e.g. historical fishing grounds for bottom contact fishing methods in moderately resilient habitat - E.g. moderate to high intensity demersal trawling
4	Partially distant to nature	- Original habitat largely removed, destroyed or permanently altered, especially where there are vulnerable and slowly recovering biota such as large and erect fauna including corals and sponges, and in areas of low productivity including deep continental slopes (depths >200 m); natural biota severely impacted or replaced by invasive or exotic species - E.g. high-intensity demersal trawl, scallop dredging
5	Distant to nature	- No resemblance to original habitat, e.g. dredged for sand or highly polluted, with no original biota or communities - Fishing influence prevents regeneration, e.g. destructive practices such as dynamite fishing
6	Non-natural artificial	- Reclaimed land with no habitat for marine species, permanent hypoxic ‘dead’ zones - No relevant fishing influence

This Table to be used in combination with Table 2. Full scale with land use descriptions in Table S2, Supplementary Information

- A measure of the total swept area of all trawls annually
- An estimate of annual average footprint based on the total area with trawl and accounting for overlapping effort within grid cells, assuming trawling is conducted randomly at sub-0.01° scale.

Three measures of area fished are compared for the seawater column case study. Area fished was calculated based on the following methods:

- 0.5° cells where fishing had been recorded over the past 10 years (Flood et al. 2014; Emery et al. 2015; SARDI 2016) with fishery-specific parameters overlaid using GIS (Table 6). This was the highest resolution available and provides a very coarse measure of area fished
- Annual area fished using data from 0.5° cells where fishing was reported for 2013
- Annual average number of net sets and gear measurements, where average purse seine net length was 1000 m, giving a radius of 159 m and an area of approximately 0.08 km<sup>2</sup> for each net set. The average number of net sets for the period was 1013 resulting in a value of 81 km<sup>2</sup> fished area.

**Table 5** Naturalness Degradation Potential (NDP) characterisation factors for seafloor and seawater column

Hemeroby class	NDP
0	0
1	0.17
<i>1.8</i>	<i>0.3</i>
2	0.33
<i>2.3</i>	<i>0.38</i>
3	0.5
4	0.67
5	0.84
6	1.0

The hemeroby scores for sardine and prawn have been italicised although these scores are not directly comparable given the different methods used to calculate area

## 2.6 Calculating catch

In the case of the Northern Prawn Fishery, catch was calculated as the average catch over the previous five years (from 2010 to 2011 to 2014–2015) as annual catch can vary substantially (Woodhams et al. 2011; Woodhams et al. 2012; Woodhams et al. 2013; Georgeson et al. 2014; Patterson et al. 2015).

Australian sardine catch is limited by quotas and the average TAC calculated for the years 2007–2015 (Ward et al. 2015) was used to correlate with area fished data. We also

**Table 6** Fishery-specific parameters

Fishery	GIS parameters	Gear and effort
Sardine ( <i>Sardinops sagax</i> )	- South Australia only - Inshore waters to the edge of the continental shelf, down to depths of 200 m - Exclude reserves and land (Flood et al. 2014; SARDI 2016)	Net length—1000 m Net sets (annual average 2007–2015)—1013 Net set area—0.08 km <sup>2</sup> (Ward et al. 2015)

used an example of yearly catch (2013) for comparison with one method of area calculation – the annual average number of net sets and gear measurements (Table 7).

### 3 Results

For each fishery case study we demonstrate several NDI scores based on the different methods for calculating area and catch described in section 2.7. The results for the different fisheries are not directly comparable given the different methods used to calculate area.

The seawater column of the South Australian Sardine Fishery was given a score of 1.8, on a scale of 0–6 where 0 characterises a natural state, using Table 1 and classified as partially close to nature (Table 3). Using the characterisation factors in Table 5, this score corresponded with an NDP value of 0.3 (Table 5). The biomass of sardines was considered to be between 30 and 74% based on model-generated estimates of spawning biomass. Schools of sardines are generally highly homogenous and the catch composition of purse seine fishing includes very little bycatch compared to other fishing methods (PIRSA 2014). There is a low level of risk to bycatch and ETP species (SASIA 2012), and retained species (other than sardines). Ward et al. (2015) found no evidence of ecological impacts from the South Australian Sardine Fishery. There was some variation in results when using annual or averaged data. The naturalness degradation value of 0.71 m<sup>2</sup> kg<sup>-1</sup> for sardine based on the average TAC for 2007–2015 was lower than the value of 0.8 m<sup>2</sup> kg<sup>-1</sup> based on 2013 catch, reflecting variability between annual and average catch data when the

method of area calculation is the same (Table 7). The three examples of area calculation demonstrated that the NDI was very sensitive to the method used, rather than the influence of the fishery. The choice of method for calculating area overwhelmed the hemeroby score and actual impact of the fishery. The naturalness degradation scores for sardine, based on the gear and effort calculation method for area, were more than two orders of magnitude lower than those based on the 0.5° cells area method. The naturalness degradation value of 365 m<sup>2</sup> kg<sup>-1</sup>, calculated using the area measurement based on fishing recorded in past 10 years in 0.05° cells, was higher than the value of 249 m<sup>2</sup> kg<sup>-1</sup>, calculated using the area measurement based on fishing recorded in 2013 in 0.05° cells. The values of 0.71 m<sup>2</sup> kg<sup>-1</sup> and 0.8 m<sup>2</sup> kg<sup>-1</sup> calculated using a measure of area based on gear and effort were significantly lower because they did not include the entire region of the GIS grid cell when only a small part of the cell had been exposed to fishing. This measure of area was most similar to the measure of total area swept in the Northern Prawn Fishery.

The seafloor of the Northern Prawn Fishery was given a score of 2.3 using Table 2 and classified as partially-close-to-nature (Table 4). Using the characterisation factors in Table 5, this score corresponded with an NDP value of 0.38 (Table 5). Trawling in the fishery occurs in resilient habitats and there is a high degree of confidence around the level of impacts with well-defined reference points established for the fishery. The naturalness degradation results varied greatly depending on the trawl area data used. The footprint area based on total area of 0.01° cells with trawling recorded in the past 5 years was almost six times larger than the footprint area that represented the annual average footprint where overlapping effort within grid cells was accounted

**Table 7** Naturalness degradation calculations for seawater column

Fishery	Hemeroby score (Table 3)	Area with fishing recorded in past 10 years in 0.05° cells (m <sup>2</sup> ) <sup>a</sup>	Area with fishing recorded in 2013 in 0.05° cells (m <sup>2</sup> ) <sup>a</sup>	Area fished based on gear/effort (m <sup>2</sup> ) <sup>a</sup>	NDP	Catch (kg)	NDI (m <sup>2</sup> /kg)
South Australian sardine	1.8	4.14E + 10	2.82E + 10	8.10E + 07	0.3	3.40E + 07 (TAC)	365
				8.10E + 07		3.40E + 07 (TAC)	249
				8.10E + 07		3.40E + 07 (TAC)	0.71
				8.10E + 07		3.05E + 07 (2013)	0.80

<sup>a</sup> Area calculated using data from Flood et al. (2014); Emery et al. (2015); SARDI (2016)

for. The NDI results varied from a score of 3.45 to 0.62 m<sup>2</sup> kg<sup>-1</sup> reflecting the different area input data (Table 8).

## 4 Discussion

The hemeroby concept has been developed here as an alternative but complementary approach to including impacts of fishing on biodiversity within LCA. This method for assessing the naturalness of marine systems has been adapted from terrestrial systems to assess the impact of fishing on the seafloor and seawater column. The developed scales are designed to facilitate comparison with land use by terrestrial food production systems, building on published studies (Fehrenbach et al. 2015; Brentrup et al. 2002) and to facilitate greater parity between assessments of marine and terrestrial food production systems. In adapting this method and applying it to our case studies, we identified a range of methodological issues that require consideration and offer some suggestions for consideration for future applications of this method.

### 4.1 Methodological issues

#### 4.1.1 Issues calculating catch and area fished

The case studies reflected the sensitivity of the NDI to the choice of area measurement. Calculating annual catches can be complicated in fisheries where catch fluctuates due to environmental or economic reasons. Variance in environmental impacts from one season to another has previously been reported by Ramos et al. (2011) and Ziegler et al. (2015). Using actual catch or TAC averaged over a number of years, with a measure of area where fishing has been reported over the past decade, was assumed to provide a robust measure of (potential) catch per area. However, results show that without a measure of actual effort, this method may seriously overstate the area fished. For example, a 0.5° cell would be included in the area measurement if it was only fished once with a very small amount of catch. The coarse spatial resolution of fishing effort data has previously been identified as a significant

problem for calculating fishing impacts (Nilsson and Ziegler 2007). Applying a lower limit of fishing effort as a criterion for inclusion of cells should be considered in the future, and application of the NDI to a wider range of fisheries is recommended to examine the influence of area at different levels of hemeroby.

The method used for calculation of area affected is clearly important when comparing naturalness degradation impacts between fisheries and also with agricultural systems, and care must be taken to ensure that the resolution is comparable. For example, land occupation data have been calculated for beef using a disaggregated land inventory and reported in square metre years (Wiedemann et al. 2015). However, the resolution of this method, and whether it considers unused areas within the land zone, is unclear. Matching the method of area calculation for beef with any of the methods used in the marine case studies was therefore not possible. If measurements of area are not compatible across systems, the assessment should not proceed to the impact assessment stage.

#### 4.1.2 Class distribution intervals and characterisation factors

There are several options for the distribution of numerical intervals between each class, including linear constant intervals, exponentially and sigmoid progressing. A linear approach was applied here following Brentrup et al. (2002); however, examining the use of a non-linear approach may be useful for future applications of this method. While the dynamics and stability of natural marine ecosystems is largely unknown, a linear response to environmental drivers has been recorded in marine ecosystems (Lindegren et al. 2016) and populations (Hsieh and Ohman 2006). However, Selkoe et al. (2015) argue that marine ecosystems tend to resist major change until they reach a tipping point. These tipping points can be quantified as zones of rapid change in a non-linear relationship between ecosystem condition and intensity of a driver. Some marine systems may be prone to tipping points, and more information is needed to identify measurable tipping points in the oceans, and the ecosystems that are likely to exhibit tipping points.

**Table 8** Naturalness degradation calculations for seafloor

Fishery	Hemeroby score (Table 4)	Area with trawling recorded in past 5 years in 0.01° cells (m <sup>2</sup> ) (Pitcher et al. 2016)	Total swept area (m <sup>2</sup> ) (Pitcher et al. 2016)	Area of annual average footprint (m <sup>2</sup> ) (Pitcher et al. 2016)	NDP (Table 5)	Catch (kg) 5-year average <sup>a</sup>	NDI (m <sup>2</sup> kg)
Northern Prawn Fishery	2.3	6.79E + 07	1.99E + 07	1.22E + 07	0.38	7.47E + 06	3.45 0.98 0.62

<sup>a</sup> Catch calculated from Woodhams et al. (2011); Woodhams et al. (2012); Woodhams et al. (2013); Georgeson et al. (2014); Patterson et al. (2015)

Characterising naturalness of systems may also be influenced by social processes, such as socio-emotional valuations of a species' functions. A social response function has been examined in relation to biodiversity offsets where the social process determining the permitted extent of ecosystem service loss over a given time horizon was modelled (Thébaud et al. 2015). Social responses to declining naturalness of systems may follow a sigmoid-shaped curve where moving from a natural state to 'close to nature' or from 'distant to nature' to artificial is more acceptable than moving between 'partially close to nature', 'semi-natural' and 'partially distant to nature'. This approach to characterisation has not previously been used in LCA that the authors are aware of but may potentially be useful in informing marine management and planning and research in the field of social life cycle assessment.

Fehrenbach et al. (2015) developed characterisation factors based on current area mix for terrestrial land use which reflected the effort required to achieve improved naturalness. While a comparable assessment of area mix does not exist for the marine environment, a global map has been developed for human impact on marine ecosystems (Halpern et al. 2008). They found that no area is unaffected by human influence, while a large fraction (41%) is strongly affected by multiple drivers. The study accounted for both marine and terrestrial impacts on the marine environment. Further classification of impacts and use in the marine environment could help to inform future development of characterisation factors.

#### 4.1.3 Individual vs collective impacts

Marine areas commonly support several distinct fisheries in terms of the target species and gear used. This situation is in contrast to the terrestrial situation where the land is more likely to be privately owned and managed, and agricultural land uses are likely to be separated spatially, for example cropping and livestock. Assessing one fishery in a multifishery zone may result in an NDI score that is not reflective of the current state of the habitat due to greater impacts from another fishery. For example, a trap fishery will have only a small impact on seafloor naturalness; however, if it is operating within a bottom trawl zone, the overall state of the area may be far from natural. One way to deal with this situation is to assess all fisheries operating in the area and score the naturalness based on the fishery with the greatest impact. Alternatively, the cumulative score of each fishery can be calculated and would represent the worst possible case. For a single-method fishery targeting multiple species, the seafloor impacts would be the same; however, seawater column scoring may vary by species. In this situation, the same approach could be applied as for the seafloor scoring, where the score is based on the species most impacted or on a cumulative score.

Another consideration is that of single species or habitats in a system that are depleted or harmed, while others in

the system are not. For example, when scoring for bycatch (Table 2), where multiple species are caught and only one of them is depleted beyond established reference points. In this situation, an average of all species could be used or alternatively the highest score of all species. Decisions made around which approach is taken to score the naturalness of an area must be identified in the goal and scope of the LCA. For example, the practitioner must define if the aim is to score the naturalness of an area based on a particular fishing method, in which case it may not represent the overall naturalness of the area, or score the naturalness of a given area based on all activities occurring there.

The sea use indicator developed by Langlois et al. (Langlois et al. 2014b; Langlois et al. 2016), which accounts for impacts from other human activities in the marine environment, uses free Net Primary Production (fNPP) to express the life support capability of the ecosystems. The use of NPP, or primary production required, is emerging as a valuable tool within LCA (Cashion et al. 2016); however, the NPP method does not capture the naturalness of systems and it is possible for managed ecosystems to have a higher 'productive value' than natural ecosystems (Taelman et al. 2016). Combining NPP with a measure of naturalness of the system can therefore provide a more holistic assessment of biodiversity impacts from fishing and other human activities. Impacts of cumulative stressors in the oceans has been identified as a top research priority (Rudd 2014), and these types of assessments will become more important as the range and intensity of sea uses increase, including uses such as marine infrastructure, coastal urban development and aquaculture facilities (Dafforn et al. 2015). A current limitation of the method presented here is that it does not capture anthropocentric changes unrelated to fisheries through midpoint indicators and needs to be combined with methods that do in order to reflect these impacts on marine ecosystems.

#### 4.2 Future application

The aim of demonstrating this method is to progress the ability of LCA to provide a measure of equivalent land use that reflects both the area fished and the extent of the damage caused and the ability of the system to recover. Further development of this method can enhance assessment of the impact of fishing within LCA to provide a measure for comparison with other fisheries and non-fished areas. Marine environments are largely opaque, and changes in marine systems are not as readily visible as in terrestrial systems. This means that causal relationships in marine environments are more uncertain than in terrestrial systems (Johnson and Sandell 2014). The scoring system presented here is novel in its attention to uncertainty. The scales can be used for fisheries where data exist, for example on resilience of habitat assemblages or where limit reference points have been established as well as for fisheries

where data are limited. Most fisheries have adequate qualitative information to enable them to be scored, although lack of data or documentation will result in higher uncertainty about the performance of the fishery (MSC 2015). For very data-poor fisheries with only catch and type of gear data, using this assessment method may be unfeasible. Greater uncertainty will result in a fishery being classified at a higher hemeroby level (further from natural) than would be the case if more information were available. Providing details on how fisheries is scored is important to ensure transparency of future assessments.

Indicators of hemeroby can also be a meaningful supplement to information provided by other national fishery indicator systems (Walz and Stein 2014). A measure of naturalness can complement established seafood sustainability and marine ecological risk assessments (ERAs) by adding another level of detail, for example where a fishery is operating at a sustainable level within a permanently altered ecosystem. A measure of naturalness may also provide additional information where current risk of habitat or ecosystem damage is considered low but is a result of prior removal of sensitive species or habitats. The degradation of naturalness of a fishery may also be an important consideration where sustainability assessments are based solely on recent data, and the assessment process may be influenced by shifting baselines. In such cases, reliance on recent data can lead to acceptance of the current situation as the natural baseline (Pauly 1995). Using recent data to calculate unfished ecological and stock baselines can be problematic, and benchmarking habitat structures (Handley et al. 2014) or drawing on other sources of historical information may be useful to better define the natural state (Pinnegar and Engelhard 2007).

In terrestrial systems, particularly in Europe, the hemeroby concept is well developed and has been used in the field of spatial planning to estimate the cumulative impact of land use changes (Walz and Stein 2014) and to help inform agri-environmental indicators developed for monitoring the integration of environmental concerns into the *Common Agricultural Policy* (European Union 2012). However, the hemeroby concept has not been applied to marine systems and, as on land, may present a useful method for spatial planning or for informing productivity/environmental indicators, such as those used in ecosystem-based fishery management. Incorporating a measure of naturalness of fished areas within planning frameworks can assist with zoning of marine protected areas and in damage assessments by informing trade-offs between development and protection. For example, assessing the naturalness of an area could help inform comparative ecosystem analyses which have been identified as effective methods for use in developing decision support tools for ecosystem-based management of marine areas (Murawski et al. 2010).

Several authors have used the hemeroby approach on the scale of terrestrial bioregions and ecoregions. Data on the type of seafloor substrate and some biome types are strongly lacking at the global scale (Langlois et al. 2016); however, a framework for classifying marine biodiversity on the seafloor has been used for continental-scale bioregionalisation (Last et al. 2010) and may provide a workable basis for defining, managing and conserving biodiversity in the sea at a global scale. Scaling up the naturalness approach in the marine environment, in combination with these types of assessments, may help inform global analysis on marine ecosystem impacts and help to prioritise management efforts to improve marine ecosystems (Halpern et al. 2015). The sustainability of seafood is also dependant on a range of socio-cultural aspects including the food provisioning functions of small-scale fisheries (Kittinger et al. 2015), resilience of fishing communities and livelihood options for current and future generations (Lam and Pitcher 2012). Conservation of marine biodiversity, therefore, needs to encompass a range of environmental parameters as well as a range of social-cultural parameters.

#### 4.3 Incorporating established frameworks into LCA

There have been calls to incorporate a life cycle approach to management and certification in seafood production for a more holistic sustainability assessment (Pelletier and Tyedmers 2008; Ziegler et al. 2016; Madin and Macreadie 2015; Hornborg et al. 2012); however, there is also merit in using existing indicators and metrics from established seafood sustainability assessment frameworks to inform the development of fishery-specific LCA indicators. A number of independent certification bodies currently assess the sustainability of wild-capture fisheries and species. The Marine Stewardship Council (MSC) is one of the more established seafood ecolabelling programs, with MSC-certified fisheries representing approximately 10% of the global harvest of wild-capture fisheries and over 19,500 products bearing the MSC label in more than 100 countries ([www.msc.org](http://www.msc.org)). The MSC's *Principles and Criteria for Sustainable Fishing* were developed through an international consultative process with fishery stakeholders (MSC 2002) and incorporate broader components of ecosystems, including the sustainability of species taken (target and bycatch), as well as the impacts of fishing on other ecologically related species; endangered, threatened or protected species; habitats and the productivity, diversity, structure and function of ecosystems (Grieve et al. 2011).

Basing the scoring system for hemeroby on established Ecological Risk Assessment for the Effects of Fishing and MSC frameworks means that the results from all three assessments will have similarities, although they measure different things, i.e. risk, sustainability or naturalness. Unlike the other frameworks, LCA provides a quantitative product assessment

and building on the well-established process for assessing fisheries within the ecosystem-based fishery management framework, as has been done here, will help to build more compatible and robust assessments of seafood products within the LCA framework.

## 5 Conclusions

Maintaining the ocean's ability to produce food for humans is important given the growing demand for protein combined with increased pressure on land and fresh water resources. Incorporating a measure of naturalness into assessments of food production can be a useful tool to better understand the cost, in terms of transforming ecosystems from natural to more artificial, of meeting the growing demand for food. The hemeroby concept has been used to assess the human impact of food production on land and may offer a useful method for assessing impacts of production in the ocean. However, a number of issues were identified in this study, including the influence of area data used and the need for comparable resolution between studies, which need further consideration for future application to assessments of marine environments.

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