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A Global LCIA Framework for Quantifying Seabed Damage from Bottom Trawling, Dredging, and Benthic Structures

Sedona Anderson,* Solveig Størdal Tveit, Konstantin Stadler, and Francesca Verones

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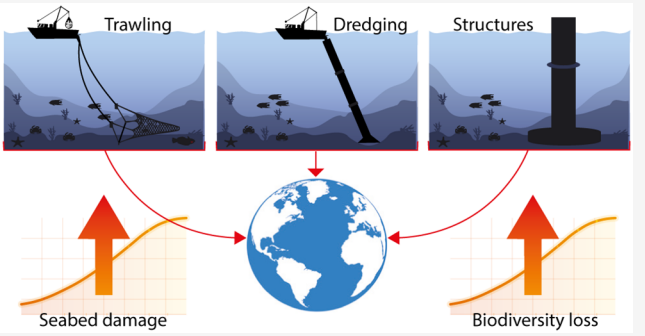
ABSTRACT: The benthic zone provides the foundation for marine ecosystem health, yet seabed damage from anthropogenic activities remains mostly unrepresented in life cycle impact assessment (LCIA). Here, we present a globally applicable LCIA characterization framework for seabed damage from three major disturbance categories: bottom trawling, dredging, and structures on the seabed. We calculate 3,800 characterization factors (CFs) across 231 marine ecoregions (MERs), including 1,900 regional CFs and 1,900 globally normalized CFs derived using global extinction probability (GEP). MER-level CFs are presented for four trawling gear types, dredging, benthic structures, as well as generic-fish and species-level trawling CFs. Species-level CFs for 51 commercially significant fish species are expressed in PDF*yr/kg, which is a functional unit directly usable by practitioners conducting life cycle assessments of seafood and marine industries. The customizable structure of the framework allows practitioners to match assessment precision to available data. Together, these advances strengthen the capacity of LCIA to account for seabed damage across the seafood, energy, and marine infrastructure sectors.



KEYWORDS: *life cycle assessment (LCA), marine biodiversity, benthic habitat, seafood sustainability, trawling, ocean impact, life cycle impact assessment (LCIA), impact model*

1. INTRODUCTION

Seabed damage is a top driver of marine biodiversity loss.[1−3] The benthic zone, or seabed, is a critical source of food and shelter for countless marine species. Benthic habitats like coral reefs and seagrass meadows provide the structure for diverse marine ecosystems.[4] These sessile benthic epifauna are home to benthic invertebrates, which feed demersal fish species.[4] Pelagic species, as well as invertebrates like scallops and oysters, require healthy benthic habitats for the survival of their juveniles.[4] Thus, the benthic zone is crucial for the overall health of the ocean, yet it suffers extensive and ongoing destruction from anthropogenic activities.[5,6]

Direct biodiversity loss from seabed damage is caused by several mechanisms: sediment abrasion from activities including bottom fishing, sediment removal from dredging, placement of hard structures on the seafloor, and smothering by sediment displacement.[7,8] In addition, indirect biodiversity loss is caused in demersal and pelagic communities due to interactions between benthic species and those in other zones.[9]

The top contributor to seabed damage is bottom trawling.[10,11] This form of benthic fishing uses weighted, cone-shaped nets that are dragged across the seafloor, causing abrasion.[1] Trawling is globally prolific, accounting for up to 25% of annual wild catch.[12] Its footprint is heterogeneously distributed, from around 10% of Australian continental shelf being affected by bottom trawling to upward of 50% of European seabed.[12] Trawling impact depends on the areal footprint, sensitivity of the habitat, and gear type.[10] Different gears have various penetration depths, which affect the intensity of the trawling activity.[10,11] Within our study, we define "bottom trawl" as any towed bottom-fishing gear including otter trawls, towed (scallop) dredges, and hydraulic dredges.

Sediment removal by dredging and construction of hard structures on the seafloor are also notable contributors to seabed damage.[1,3] Dredging is the excavation of benthic material (sediment) using suction or mechanical equipment to remove, transport, and deposit the material at a new location.[13] It may be conducted to improve navigational access or to remove material for use in other locations.[14] Dredging removes the entire benthic layer, causing potential changes to bathymetry and erosion in the surrounding ecosystem, as well as reducing light availability, increasing turbidity, and

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changing the water quality.[14−16] Hundreds of millions of cubic meters of sediment are dredged annually,[17] and demand for marine aggregates like gravel and sand continues to grow with global populations and infrastructure needs. Between 2000 and 2012, turnover for dredging contractors worldwide doubled to 11,370 million euros.[13]

Structures on the seabed remove the entire benthos where they are constructed and prohibit benthic recovery as long as the structure is in place. Within our study, this category encompasses structures related to a variety of activities including renewable energy foundations, oil and gas drilling foundations, and underwater power cables. Impacts come from the removal and obstruction of the seabed.[18−20] This category of activities is more nuanced to evaluate as structures can also have a net-positive impact on biodiversity through the cessation of fishing activity, which may shift impacts elsewhere, or artificial reef effects.[18] As energy demand continues to grow, so does benthic disturbance due to structures for energy infrastructure, and quantifying the impact of benthic disturbance from this and the other aforementioned activities is critical for preserving ocean health.

Life cycle impact assessment (LCIA) is an analytical framework that can be used to achieve this by quantifying the potential environmental impacts of human activities and identifying leverage points for mitigation. However, compared to terrestrial systems, marine impacts are still underrepresented in LCIA modeling.[21] This gap constrains the ability to perform robust environmental assessments that are crucial for guiding policy and regulation toward sustainable ocean management.[1,22] Expanding the LCIA framework to include the ecological consequences of seabed damage strengthens the capacity of LCIA to inform sustainable marine management and ensure that decisions about ocean use account for seabed health.

Prior efforts to bring seabed and sea-use damage into LCIA have taken different approaches. Langlois et al. (2015) developed CFs for sea use that quantify impacts on marine "life support functions" via biotic production (kg C equivalent),[23] while Préat et al. (2021) proposed a framework using the indicator potentially disappeared fraction (PDF) of species for deep seafloor ecosystems, applied to nodule mining in the Clarion−Clipperton Zone.[24] These differ from the present work in end point and/or spatial scope. We express seabed damage using PDF, the standard ecosystem-quality end point used across mainstream LCIA methods.[25] The most directly comparable predecessor is the characterization model of Woods and Verones (2019), which quantifies seabed damage as PDF of species per m^2 of seabed disturbed and provides CFs for 17 European marine ecoregions.[8] However, several research gaps remain; the model's regional focus limits its global applicability, the CFs are irrespective of activity, and the functional unit (m^2) means that it is largely theoretical—at least for trawling—as practitioners would be unlikely to have this information.

With our study, we present a new seabed damage model with global coverage and improved ecological resolution that includes characterization factors for trawling, dredging, and structures on the seabed. Smothering, which follows a different impact pathway,[8] and indirect biodiversity loss are outside the scope of this study. Critically, for trawling we provide CFs in PDF per kg fish trawled for 51 species, as well as values for generic fish, enabling practitioners to conduct meaningful LCIA assessments. These novel developments enhance the robustness and policy relevance of characterization factors for seabed damage to be used in LCIA.

2. METHODOLOGY

2.1. Spatial Delineation

We quantify characterization factors for impacts from three different activities causing benthic habitat loss: trawling, dredging, and structures on the seabed. Each set of characterization factors is spatially delineated at the marine ecoregion (MER) level.[26] Marine ecoregions are the units of the Marine Ecoregions of the World (MEOW) system, a global biogeographic classification that divides the world's coastal and continental-shelf waters into 232 areas of relatively homogeneous species composition, each distinguished from adjacent areas by its species assemblage and oceanographic or topographic setting.[26] They are the marine equivalent of the terrestrial ecoregions that serve as a spatial unit in biodiversity-related LCIA[27] and they provide a natural and internationally consistent basis for marine CF development.[26] We considered calculating CFs for open ocean areas, however we found that majority of the considered activities occur in coastal/shelf areas, with the maximum depth being 2000 m.[12,28−30] The activities are therefore well-covered by MERs and considering that habitat and substrate data for the open ocean is very sparse, we chose to calculate CFs only at the MER level. This represents one of several possible granularities at which marine biodiversity could be delineated and because our CFs are expressed as a PDF per unit area, this choice influences their absolute magnitude, a point we return to in section 3.1.

When calculating MER areas, each spatial unit was quantified in m^2 using geodesic calculations on the ellipsoid in EPSG:4326. Terrestrial areas of MERs were removed before conducting area-based calculations, which were completed using ArcGIS Pro version 3.3.0,[31] R version 4.5.2,[32] RStudio 2025.09.2, and Python version 3.13.0.

2.2. Initial Disturbance Impact

All variables and their definitions can be found in SI1, table S1. The initial disturbance D (eq 1), differentiated for activity i and ecoregion j, reflects the immediate benthic response following a disturbance. It is determined by the product of the initial species loss P of activity i and the inverse of the area A of the affected ecoregion, j:

$$D_{i,j} = P_i \times \left(\frac{1}{A_j}\right) [\mathrm{PDF/m^2}] \quad (1)$$

P_i expresses the potentially disappeared fraction (PDF) of species that is associated with a given activity's disturbance intensity (Table 1). For both dredging and structures on the seabed, the entire benthos is removed, thus P_i is equal to 1. For trawling, the intensity is dependent on what type of gear is used. Sciberras et al. (2018) published a meta-analysis of 122 experiments on the "effects of bottom fishing to quantify the removal of benthos" with rates given in loss of species richness per gear pass, which we have used as rates for trawling P_i values. Where gear type is unknown, we provide an arithmetic mean of the gear-specific intensities. Though considered, a use-weighted value is precluded because no data set resolves trawling

Table 1. P Values by Activity i.[a] [33]

	Activity (i)	P
Trawling	All gear types, arithmetic average	0.18
	Otter trawl (OT)	0.09
	Towed dredge (TD)	0.12
	Hydraulic dredge (HD)	0.32
	Dredging	1.00
	Structures on the seabed	1.00

[a]P indicates the fraction of species removed from the benthos [PDF] per gear pass or action (i.e., dredging, structure placement).

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activity into these gear categories at the global scale. Descriptions of the gear types are available in SI1, table S2.

2.3. Ecological Recovery Time

After the initial disturbance, the ecological recovery period begins. Recovery time T (eq 2) for each ecoregion j is calculated as defined in Woods and Verones (2019):[8]

$$T_{s,j} = M_s \times H_j \times R_{s,j} \text{ [yrs]} \quad (2)$$

The recovery time is comprised of the median recovery time M_s [yrs] for substrate s which is either lengthened or shortened by the two factors H_j [-], hydrodynamic energy level at the seabed, and $R_{s,j}$ [-], the stock of potential recolonizing organisms.

2.3.1. Median Recovery Time (M_s). M_s values are specific to each seabed substrate type. Woods and Verones (2019) report median recovery times per substrate for temperate and polar regions and we derived corresponding medians, where possible, for tropical and subtropical zones from a meta-analysis.[34] For muds and sands, we combined the temperate/polar and tropical/subtropical medians into a globally representative value per substrate by taking their arithmetic mean, weighting each climate zone equally. We chose to weight the zones equally rather than pooling all observations into a single median due to fewer tropical/subtropical data points, which would have resulted in a median value dominated by temperate and polar data. For rock and biogenic reef and for coarse or mixed sediments, no tropical or subtropical recovery-time studies exist, so the Woods and Verones (2019) values were used directly.

A global map of seabed substrate types was sourced from Dutkiewicz et al. (2015).[35] To account for variable naming conventions, the substrates from the global data set needed to be aggregated to the categories in Table 2. An overview of this aggregation can be found in SI1, table S3.

Table 2. Substrate Classifications and Representative Recovery Times M_s [Yrs] per Substrate Type s[a]

Substrate type (s)	M
Rock or biogenic reef	18.50
Coarse or mixed sediments	5.50
Sands	3.05
Muds	0.95

[a]Values for muds and sands are the arithmetic mean of temperate/polar and tropical/subtropical median recovery times ; values for rock or biogenic reef and for coarse or mixed sediments are the median recovery times reported by Woods and Verones (2019) See Section 2.3.1 for derivation.

2.3.2. Hydrodynamic Energy Level (H_j). Hydrodynamic energy H_j affects recovery time as shallow areas typically experience greater natural disturbance that aids in regeneration, while deeper areas experience less natural disturbance and a lower ability to quickly recover.[36] M_s values are assumed to represent recovery under moderate hydrodynamic energy, as determined by Woods and Verones (2019), and are modified by H_j to account for regional variation. Given a lack of global data, depth was used as a proxy for hydrodynamic energy.[8] Sea depth values were taken from the GEBCO 2024 bathymetry grid.[37] Hydrodynamic energy classes based on depth are sourced from Woods and Verones (2019) (SI1, table S4). Using a raster of bathymetry values masked by an ecoregion j, each pixel in j was given a corresponding hydrodynamic energy class value based on depth. Subsequently all pixel values were averaged to compute a single, regional H_j value.

2.3.3. Potential Stock of Recolonizers ($R_{s,j}$). The potential stock of recolonizers R, for substrate s and ecoregion j, is a factor where we consider the number of habitats on a substrate to represent the likelihood of recolonizing organisms (eq 3).

$$R_{s,j} = \frac{\tilde{F}_s}{F_{s,j}} \text{ [-]} \quad (3)$$

$F_{s,j}$ is the fraction of region j composed of unique habitats on substrate s (eq 4), and $\tilde{F}_s$ is the median of $F_{s,j}$ across all regions. Thus, if a region has a value that is lower than the median, it extends the recovery time, while a higher value shortens it. $F_{s,j}$ is calculated as

$$F_{s,j} = \frac{A_{s,j}}{A_j} \times \frac{1}{N_{s,j}} \text{ [-]} \quad (4)$$

To calculate eq 4, we use the total area A of substrate type s within region j. However, the global substrate data set has a latitudinal extent of −89.95 to 86.95 degrees and is therefore missing some information at polar latitudes. For the affected regions, we took the total available substrate area and the proportion of that area belonging to each substrate type, and using those proportions scaled the substrate areas to the total region area. There are 18 affected polar ecoregions, of which 7 have fewer than 50% of the total area substrate mapped. One ecoregion (the High Arctic Archipelago) has no mapped substrate and has been excluded. The affected ecoregions and the percentage of their area substrate that is assumed can be found in the SI1, table S5.

$N_{s,j}$ represents the number of unique habitats on substrate s in region j. Detailed habitat data is not available globally. Therefore, habitat density functions for each substrate type were used to estimate the number of habitats in regions where detailed habitat data was not available. Habitat counts per substrate type were compiled from regional data sets: EMODnet for Europe, the Caspian, and Caribbean Seas,[38] and Seamap Australia for tropical regions.[39] The relationship between habitat count and substrate area was fitted using power-law regressions, yielding habitat density functions y_s (m^{-2}) specific to substrate s, which were then used with the areas of MERs to estimate the number of habitats per substrate for each ecoregion j. The regression equations can be found in SI1, section 06.

The total number of habitats N per substrate s in area j was then obtained as (eq 5):

$$N_{s,j} = y_s \times A_{s,j} \text{ [-]} \quad (5)$$

2.3.4. Calculating Regional Recovery Time. Using eqs 3-5, along with M_s and H_j, final recovery times ($T_{s,j}$) can be calculated per substrate s in region j, and then aggregated to the region level using an area-weighted mean (eq 6):

$$T_j = \frac{\Sigma(T_{s,j} \times A_{s,j})}{A_j} \text{ [yrs]} \quad (6)$$

2.4. End Point Characterization Factors

We can now calculate end point CFs (eq 7) for each activity (eq 8) by multiplying the initial disturbance (D), the ecological recovery time (T), and a factor of 0.5 to account for the linear recovery of species richness as described in Woods and Verones (2019):

$$CF_{i,j} = D_{i,j} \times T_j \times 0.5 \text{ [PDF*yr/m}^2\text{]} \quad (7)$$

The CFs are specific to activity i, and ecoregion j.

2.4.1. Trawling. For trawling, practitioners are much more likely to conduct an LCA for a given amount of catch, rather than knowing the area trawled. Therefore, we link the CF to fish using catch per unit area (CPUA) data.

Maureaud et al. (2024)[40] provide a comprehensive data set of fish biodiversity from bottom-trawl surveys. The data set provides mean weight CPUA values [kg/km^2] for 2,170 fish taxa from 216,548 hauls spanning 1963 to 2021. All fish in the data set would be caught with otter trawls, thus we use OT for the intensity (P_i) value (see Table 1). We chose the year 2000 as our temporal cutoff and filtered out any earlier hauls, leaving 1,930 fish species. We removed all species without dedicated trawling fisheries. The CPUA values for the 215 remaining species were arithmetically averaged to use for calculating a "generic fish" CF, while 103 species deemed moderately to highly economically significant were selected for individual species-level CFs.

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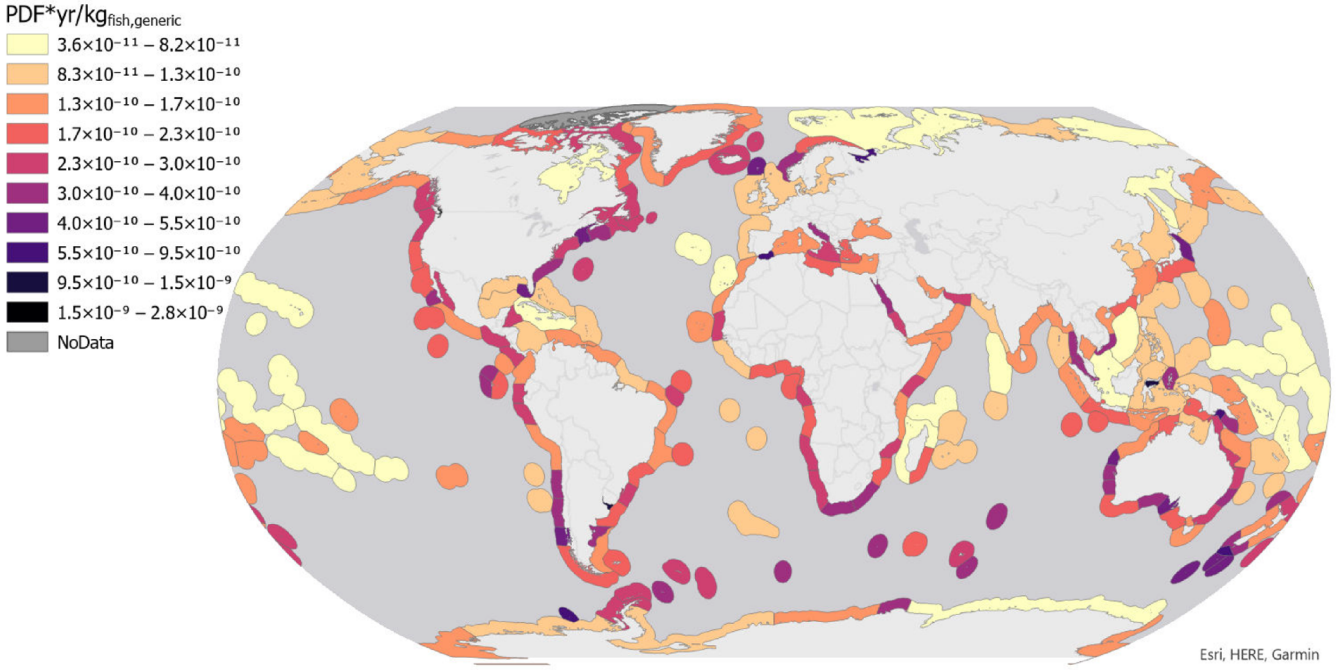

Figure 1. CFs per MER for generic fish [PDF*yr/kg], classified using Jenks natural breaks. Continent basemap from ESRI.[44]

These are calculated using an area-weighted average of the MER CFs produced using eq 7.

We sourced species range maps from the IUCN (International Union for Conservation of Nature) (2026).[41] The range maps serve as the spatial weights used to aggregate ecoregion-level CFs to the species level, where each species' range polygon is intersected with the marine ecoregions and the CF of each intersected ecoregion is weighted by the area that the species' range occupies within it. The species-level CF (eq 8) is therefore obtained as a range-area-weighted mean of the ecoregion CFs from eq 7:

$$CF_{i,p}^{m2} = \frac{\sum_j (CF_{i,j} \times A_{p,j})}{\sum_j A_{p,j}} \text{ [PDF*yr/m}^2\text{]} \quad (8)$$

Where $CF_{i,j}$ is the ecoregion-level CF for activity i and ecoregion j (eq 7), $A_{p,j}$ is the area of overlap between the IUCN range polygon of species p and ecoregion j, and the summation runs over all ecoregions intersecting the species' range, so that $\sum_j A_{p,j}$ is the total mapped range area of species p.

As this aggregation requires a range map to supply the weights, a species-level CF can only be produced for species with available range data. Of the 103 species selected, the IUCN had range maps for 51 of them. We can then use the CPUA values, along with the area-weighted species CF [PDF*yr/m^2] to calculate final CFs for species p (eq 9):

$$CF_{i,p}^{kg} = \frac{(CF_{i,p}^{m2} \times 10^6)}{CPUA} \text{ [PDF*yr/kg]} \quad (9)$$

2.4.2. Dredging. Impacts from dredging are calculated at the marine ecoregion level using eq 7 and an intensity (P_i) value of 1, due to the entire benthos being removed.

2.4.3. Structures on the Seabed. The intensity (P_i) value for structures on the seabed is 1, due to the entire benthos being removed. Whereas the initial disturbance of trawling or dredging happens nearly instantaneously and then ecosystem recovery begins, structures on the seabed persist for years or decades, preventing recovery, and thus require a slightly different methodology. Equation 1, the disturbance impact (D), can be used as a standalone CF multiplied by the years that the structure remains on the seabed to represent the "occupation" impact ($CF_{occ,j}$). Equation 7 is used to quantify the impact once the structure has been decommissioned and the "transformation" back to a natural state has begun ($CF_{trans,j}$). This aligns with a common LCIA approach for quantifying impacts from land use change.[42] The two CFs (eqs 1 and 7) can be used separately or added together to represent total impacts from disturbance through recovery.

2.4.4. Global Extinction Probability. The CFs described above express seabed damage as a regional PDF: the potentially disappeared fraction of species relative to each marine ecoregion's own species pool. Expressed in this form, impacts in different ecoregions are not directly comparable as regional and global species loss can differ substantially. Whereas a cosmopolitan species lost from one ecoregion may exist elsewhere, an endemic species lost regionally may be irreplaceable. To place impacts on a common global scale, we applied the global extinction probability (GEP) scaling approach of Kuipers et al. (2019), which uses species range sizes, global conservation status, and species richness to quantify the extent to which regional species loss contributes to global species loss.

Following Verones et al. (2022), a single multitaxa marine GEP value was derived for each of the 231 marine ecoregions, using their provided methodology.[43] Of the marine taxonomic groups for which GEP values were available, we combined those that are benthic or predominantly benthic and therefore most directly affected by seabed damage: cnidarians, holothurians (sea cucumbers), lobsters, and cartilaginous fishes. The cartilaginous fish group was included because a large proportion of the group (skates, rays, chimaeras, and many demersal sharks) live in close association with the seabed. Conversely, ray-finned fish (Actinopterygii) were excluded. It is the most speciose group and is highly heterogeneous, including many pelagic and demersal species. Though some ray-finned species are benthic, a single class-level GEP would include a large number of species with little or no association to the seabed.

Each ecoregion-level CF was multiplied by the GEP value (eq 10) of the corresponding ecoregion to obtain a CF representing global PDF:

$$CF_{global,i,j} = CF_{i,j} \times GEP_j \text{ [PDF*yr/m}^2\text{]} \quad (10)$$

where GEP_j is the dimensionless GEP factor for ecoregion j. As GEP is dimensionless, the unit is unchanged and only the scope of the fraction shifts from regional to global. For species-level and globally aggregated CFs, GEP was applied at the ecoregion level prior to range-area-weighted aggregation (eq 8).

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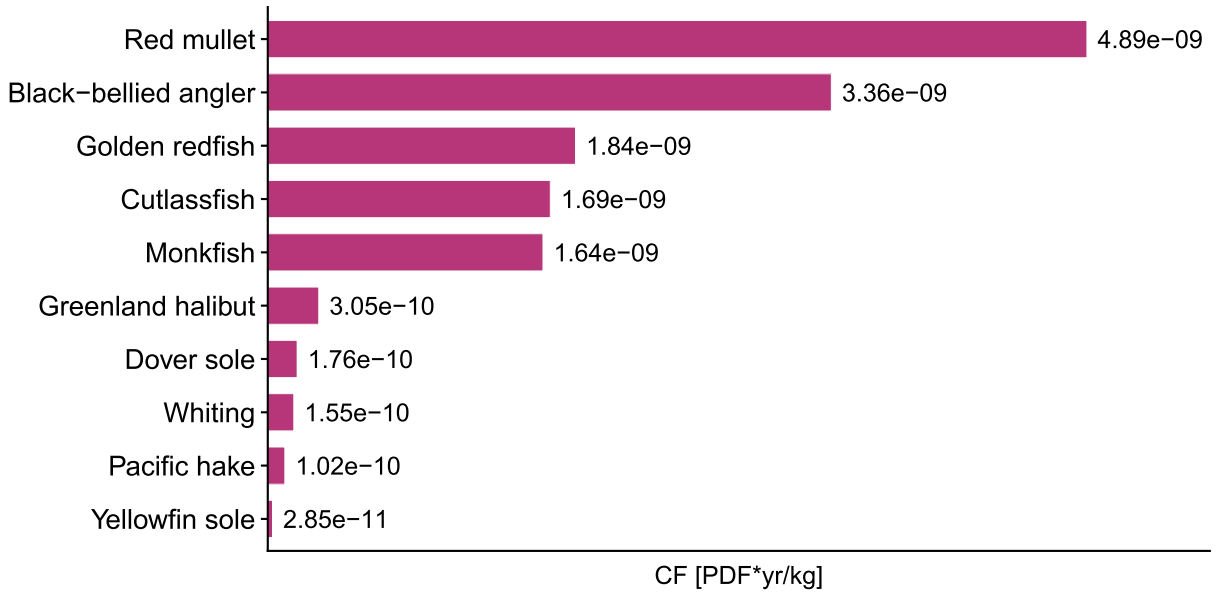

Figure 2. CFs per target species [PDF*yr/kg] for 10 of the most economically significant species by catch volume.

3. RESULTS AND DISCUSSION

3.1. Trawling

We calculated 924 regional CFs at the MER level in the unit PDF*yr/m^2 including 231 CFs for each of the 4 different trawling gear types. There are 231 CFs at the MER level in the unit PDF*yr/kg generic fish, and 51 CFs at the individual species range level in the unit PDF*yr/kg species (SI2; tables S10−12). The CFs for generic fish at the MER level range from 3.63×10^{-11} PDF*yr/kg to 2.74×10^{-09} PDF*yr/kg (Figure 1). The globally aggregated, area-weighted CF for generic fish is 1.47×10^{-10} PDF*yr/kg.

Area (A_j) is the most influential parameter for CFs at the MER level. There is an inverse relationship between area and impact that has a methodological implication which should be considered during interpretation. The initial disturbance is expressed as PDF per unit area, and therefore each CF scales inversely with the area of its MER. The MER delineation, though biogeographically justified in section 2.1, represents one of several defensible levels of granularity; a coarser or finer regionalization would shift all CF magnitudes correspondingly. This reflects the intended logic of a regional PDF such that a square meter of disturbance is a smaller fraction of the species pool of a larger region than that of a smaller one. This is insignificant for relative comparisons at a fixed resolution, where all CFs shift together, but becomes consequential when PDF*yr values are aggregated with other categories at the AoP. The CFs therefore depend not only on the causal model linking disturbance to biodiversity loss, but also on the choice of spatial reference, which practitioners should keep in mind when interpreting absolute magnitudes. Additionally, though these factors share the PDF*yr unit of other ecosystem-quality methods, marine and terrestrial impacts rest on different species pools and disturbance mechanisms and should be assessed separately rather than aggregated across realms. These considerations apply to CFs generated for trawling, but also for dredging and structures on the seabed.

For trawling, two notable areas with the most severe impacts are the Puget Trough/Georgia Basin and the Rio de la Plata MERs, which are also the smallest regions by area. Many of the midpacific MERs are among the largest by area and accordingly have some of the smallest CFs. Among MERs of similar area, spatial heterogeneity arises largely from differences in substrate composition and the resulting recovery-time weighting. A sensitivity analysis for all activities is available in SI1, table S8.

At the species level, the CFs range from 2.85×10^{-11} PDF*yr/kg for *Limanda aspera* (Yellowfin sole) to 2.04×10^{-08} PDF*yr/kg for *Dicologlossa cuneata* (Wedge sole). Figure 2 shows the comparative impacts of 10 economically significant species.

The parameter influence at the MER level also applies at the species level, but the most influential parameter is now the CPUA. This parameter has an inverse linear relationship to the CF, meaning fish that either have a high body mass or are especially abundant have a higher CPUA, and thus a lower CF as it requires less disturbed area to acquire 1 kg of catch, and vice versa.

The CFs for trawling are easily customizable based on how much data a practitioner has—the better the data, the higher the accuracy they can achieve. At the coarsest level, if a practitioner does not know the target species or location, they can use the global CF for generic fish. If they know the fishing location but not the target species, they can use the spatialized generic fish CFs. If the target species is known, they can use the species level CFs. These are quantified using weighted-average species range data and averaged CPUA data. Because each species-level CF incorporates the area trawled to land that species (its species-specific CPUA), these CFs are intended for assessing the catch of a single target species. Impacts should not be summed across multiple species caught over the same ground to assess a mixed or multispecies fishery, as each species-specific CPUA embeds that ground's disturbance independently and combining them would double-count the shared trawl passes. To calculate the most precise CFs, a practitioner would need to know the target species, know the location of the fishing activity, and have localized CPUA data. With this information, they can use the appropriate MER CF from SI2 in PDF*yr/m^2 and eqs 8 and 9 to calculate the finest resolution CF at the species level.

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When customizing CF values, practitioners should be aware that different gear types are used for different taxa (examples given in SI1, table S2). They should thus select the corresponding gear-specific MER CF from SI2, tables S10/S13. The all-gear arithmetic-average CF is intended only as a fallback for cases in which the gear type is unknown and cannot be assumed using the target taxa. As it weights the three gear types equally, and because those gears target largely nonoverlapping resources, it should not be interpreted as representing a realistic gear mix or any specific fishery. Where possible, the matching gear-specific value should always be preferred.

There are a number of ways that the trawling CFs calculated in this study could be expanded upon in future research or by individual practitioners. Though we had CPUA data for 215 trawled species, we limited the number of species level CFs first by economic relevance, and then by available range data from the IUCN. Thus, the number of species-level CFs could be increased by including species with low economic relevance, or through the acquisition of more species range data, either from additional sources or the interpolation of point data.

Within our study, we have modeled a single-impact perspective meaning, in theory, a disturbance upon a pristine state. However, bottom trawl fisheries often disturb the same area cyclically. If the interval between disturbance events is long enough that the ecosystem can achieve full ecological recovery, then the single-impact model is sufficient for characterizing impacts. However, trawling also reoccurs in areas that are not fully recovered. In this case, a single-impact perspective does not fully capture the impacts. Woods and Verones (2019) introduced a framework for a repeated-impact perspective. However, due to data uncertainty that prevents this framework from being operational at a global scale, we have chosen to exclude repeated-impact CFs from our scope. In future research, the Woods and Verones (2019) framework could be used as a starting point for global quantification of repeated impacts, which would create a more holistic view of impacts from activities like trawling.

3.2. Dredging

For dredging, we have calculated 231 regional CFs at the MER level in the unit PDF*yr/m^2. All dredging values can be found in SI2, table S10. The CFs range from 3.09×10^{-13} PDF*yr/m^2 to 2.34×10^{-11} PDF*yr/m^2. The distribution pattern of impact severity is the same as what is represented in Figure 1.

For CFs to have real-world applicability, it is important that they are expressed in the functional unit that a practitioner will use when conducting a damage assessment. In the case of dredging, that might be 1 kg of marine sand or gravel, for example. Due to data and scope limitations, our CFs are presented in PDF*yr/m^2. However, they could be converted to PDF*yr/kg aggregate using industry data that links dredged area to mass of material extracted. The conversion requires two parameters: an estimate of the volume of sediment removed per unit area [m^3/m^2], equivalent to the effective dredge cut depth, and the bulk density of the sediment [kg/m^3]. Combining a cut depth estimate with the bulk density yields a mass per unit area [kg/m^2], by which the provided CFs in PDF*yr/m^2 can be divided to obtain PDF*yr/kg aggregate.

3.3. Structures on the Seabed

For structures on the seabed, we have calculated 231 regional CFs at the MER level for both occupation [PDF/m^2] and transformation [PDF*yr/m^2] impacts, as described in section 2.4.3. All values can be found in SI2, table S10. For occupation, the CFs range from 3.77×10^{-13} PDF/m^2 to 6.31×10^{-11} PDF/m^2. For transformation, the CFs range from 3.09×10^{-13} PDF*yr/m^2 to 2.34×10^{-11} PDF*yr/m^2.

The interpretation of LCA results for structures on the seabed should be conducted with extra care, as structures can introduce other conditions that may have a positive effect on ecosystems under certain circumstances. Offshore energy infrastructure in particular may induce a cessation of bottom trawling within exclusion zones, allowing previously disturbed benthos to recover.[18] Additionally, hard structure foundations and their associated scour protection can function as artificial reefs, providing novel substrate for colonization and potentially enhancing local species abundance and diversity, especially in otherwise homogeneous soft-sediment environments.[19,45] These positive externalities are outside the scope of the present CFs, which capture only the direct negative impact of seabed occupation and subsequent transformation, but they should be considered by practitioners when interpreting results, at least qualitatively.

It is also important that practitioners apply the CFs to the correct areal footprint. The relevant footprint is the total occupied seabed area including primary foundations, but also any anchor systems, cable routes, or scour protection. Beyond seabed occupation, structures on the seabed can cause additional externalities outside the scope of this study: the introduction of artificial hard substrate into soft-bottom ecosystems, smothering from sediment displaced during construction, sound and light pollution, anchor scarring, and corrosion and chemical leakage from structures.[18−20]

3.4. Global-Scale CFs

For all 1,900 regional CFs, we have applied ecoregion-specific GEP factors to convert the regional CFs into globally comparable values, expressed as global PDF*yr/m^2 and global PDF*yr/kg. These can be found in SI2, Tables S13−S15. As GEP differs among ecoregions according to the global extinction relevance of their biota,[46] the global CFs rerank ecoregions relative to the regional values. The ecoregions with the highest global CFs (East China Sea, Central Kuroshio Current) therefore differ from those with the highest regional CFs (the Puget Trough/Georgia Basin and Río de la Plata), illustrating that GEP redistributes rather than uniformly rescales impact.

The regional and global CFs serve different purposes. The regional CFs express seabed damage relative to each ecoregion's own species pool and are appropriate for comparisons within the marine realm at a fixed spatial resolution. The global CFs place these impacts on a common basis with other ecosystem-quality impact categories, following the scaling logic of Kuipers et al. (2019), and are the appropriate choice when seabed damage is considered alongside other marine pathways within the ecosystem-quality AoP. We provide both scales to enable practitioners to match the CF to the scope of their assessment.

One caveat applies specifically to the globalized CFs. The GEP scaling is derived from four benthic or predominantly benthic taxonomic groups (cnidarians, sea cucumbers, lobsters, and cartilaginous fishes) selected because they are the most directly exposed to seabed disturbance among the groups for which GEP values were available. The resulting global CFs therefore express the contribution of regional seabed damage to global extinction risk as estimated from these groups, rather

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than from the full community affected by seabed damage. Taxa that are affected but not represented in the GEP weighting do not contribute to the scaling. The regional CFs, by contrast, are based on overall benthic species richness and do not carry this taxonomic restriction. Practitioners should therefore treat the global CFs as a globally comparable but taxonomically bounded expression of seabed-damage impact, and may prefer the regional CFs where a broader richness measure is more appropriate to the assessment. A full discussion of the uncertainties and limitations of the GEP scaling factors used here can be found in Verones et al. (2022).

3.5. Limitations

Several model parameters warrant discussion with respect to the uncertainty they introduce to the final CFs. The intensity (P_i) of seabed disturbance is influenced by two main factors: the type of gear being used and the composition of the substrate. In the predecessor framework,[8] P_i values were sourced from a study of only one gear type. Our work provides a significant improvement in including 6 different intensity values. However, we consider P_i to be spatially generic which does not reflect real world conditions. Certain studies show a linear correlation between substrate gravel content and species loss from disturbance events.[11] Future research could reduce uncertainty by spatially differentiating P_i values, defining them by both gear type and substrate.

The median recovery time values M_s also carry uncertainty, particularly for tropical/subtropical regions as a result of data deficiency. Woods and Verones (2019) report M_s for temperate and polar regions only, so we chose to derive tropical/subtropical medians from an older meta-analysis to improve regional representation.[34] Several of the underlying tropical/subtropical recovery estimates are bounded by the duration of the source experiments rather than representing a fully resolved recovery trajectory, meaning our derived medians may understate true recovery time for the affected substrates. We were unable to find a more recent, tropical-specific alternative. There is a more comprehensive global meta-analysis of experimental trawling recovery,[33] however it resolves recovery time by gear type and taxon and does not stratify by climate zone or region, so it could not be used to update the tropical/subtropical values. As a plausibility check, our blended M_s values fall within the global range of recovery times reported from large-scale comparative studies of trawling,[11] and the relative ordering across substrate types is consistent with substrate-specific recovery patterns reported elsewhere.[47] Neither of these studies resolve recovery by both substrate and climate zone, so this comparison should be read as evidence of general consistency rather than independent validation of the tropical/subtropical estimates specifically.

We quantified the propagation of parameter uncertainty for trawling parameters P_i and M_s into the CFs using a Monte Carlo simulation for a representative ecoregion (SI1 08). CF uncertainty is dominated by the initial species-loss and is largest for otter trawling, whose initial depletion is not statistically distinguishable from zero, and recovery-time uncertainty adds a further, smaller component. We emphasize that this analysis addresses parameter uncertainty only. It does not capture the representativeness uncertainty involved in applying globally compiled, predominantly temperate in situ data to specific ecoregions, which serves as a qualitative but likely substantial source of uncertainty.

Species-level CFs are currently provided only for fish, yet shellfish represent a major target of benthic trawling. While robust CPUA data exists for several shellfish species, the absence of IUCN range data for marine invertebrates has prevented their inclusion. This is a notable omission, as shellfish are targeted using more destructive gear types than the otter trawls used for fish,[33] meaning that broad assessments relying solely on fish CFs risk underestimating the full impact of trawling activity. Should more extensive invertebrate range data become available, shellfish CFs can be readily calculated using the framework presented here.

Additionally, high-resolution seabed data does not exist at the global level, particularly for habitats and to a lesser degree substrates. We use what habitat data is available, which is for Europe, the Caspian and Caribbean Seas, and Australia, and extrapolate the habitat-substrate relationships present in these regions to the rest of the world. While these regions represent a large share of marine ecosystems, there are unique areas where this extrapolation is likely to introduce a significant uncertainty, such as polar ecosystems and deep-sea benthic zones. As detailed in the methodology, substrate data only reaches −89.95 to 86.95 degrees, and therefore polar regions require a greater degree of extrapolation than nonpolar regions. Additionally, substrate areas in deeper waters are interpolated from sparser point data, and deep-sea benthic habitats are sufficiently different from coastal environments that applying coastal habitat relationships to these areas is likely to skew impact results. Despite these limitations, the model is well-grounded in empirical data for coastal and shelf environments, where nearly all commercial seabed disturbance activities occur.

The recolonization factor $R_{s,j}$ introduces uncertainty through its treatment of habitat density as a proxy for recolonization potential. By comparing habitat density on a given substrate type to the global median, the method uses habitat extent as a proxy for the stock of potential recolonizers, on the assumption that greater habitat availability within an ecoregion corresponds to a larger pool of source populations from which a disturbed area can recover. However, this approach does not account for the taxonomic or functional composition of neighboring communities. This distinction is ecologically significant as postdisturbance recolonization is not taxonomically neutral. Opportunistic, short-lived, and mobile taxa are at an advantage, while sessile, long-lived organisms are disproportionately slow to recover.[33] The model may therefore overestimate recovery by treating rapid recolonization by opportunistic taxa as equivalent to full community recovery. In areas subject to chronic disturbance, this dynamic can drive a progressive shift in community composition, favoring some organisms at the expense of the slower-recovering species that characterize undisturbed benthic ecosystems.

Lastly, the use of IUCN species range maps to weight and aggregate ecoregion-level CFs to the species level introduces uncertainty related to the known limitations of these data in the marine realm. Marine ranges are often drawn as broad, continuous polygons that can overestimate true presence by implying uniform occupancy across ecologically heterogeneous areas. This means that ecoregions where a species is rarely or marginally present may be weighted similarly to those where it is genuinely abundant, potentially skewing the species-level CF away from the areas of greatest ecological relevance. This limitation is most pronounced for species with poor observational coverage, which are disproportionately repre-

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sented in the deep sea and polar regions—the same areas identified above as carrying the highest spatial uncertainty for other model inputs.

In total, these uncertainties are acknowledged as inherent to a globally applicable model; the cumulative effect is greatest in data-poor regions such as deep-sea and polar environments, while for the coastal and shelf areas that represent the primary domain of seabed-disturbing industries, the model relies on strong empirical foundations.

3.6. Implications

This study presents the first globally applicable LCIA characterization framework for seabed damage across three major categories of anthropogenic disturbance: bottom trawling, dredging, and structures on the seabed. By expanding from the 17 European ecoregions of the predecessor model to 231 marine ecoregions worldwide, and by introducing CFs in practitioner-relevant functional units, specifically PDF*yr/kg for 51 individual fish species and a generic fish category, this work represents a methodological step forward in the capacity of LCIA to represent marine impacts. The customization potential of the trawling CFs allows practitioners to match assessment precision to available data, from a global generic value down to a species and location specific CF, making seabed damage characterization practically accessible in a way that has not previously been possible. The explicit inclusion of dredging and structures on the seabed broadens the scope of activities that can be assessed, reflecting the diverse and growing set of industries interacting with the benthic zone.

The model is intended to provide a transparent and globally consistent basis for quantifying seabed damage impacts where no such tool previously existed. As global marine monitoring efforts improve, particularly with respect to deep-sea data, polar substrate data, and species occurrence records, the framework presented here is explicitly designed to accommodate updates, whether by the broader research community or by individual practitioners with access to localized data. The ocean is among the least understood environments on earth and faces increasing and synergistic anthropogenic pressures, and equipping the LCA community with tools for quantifying human impacts is a prerequisite for making it visible in the decisions that matter.

■ ASSOCIATED CONTENT

Supporting Information

- The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.est.6c10111.

SI1: Table S1, variables used in the characterization framework and their definitions; Table S2, description of fishing gear types used for trawling P_i values; Table S3, alignment of substrate classification systems used in CF modeling and Dutkiewicz et al. (2015); Table S4, depth ranges for assigning hydrodynamic energy at the seabed; Table S5, polar ecoregions with incomplete substrate coverage and estimated substrate proportions; Section 06, habitat density power-law regression equations; Table S6, fit statistics for the habitat density regressions; Table S7, habitat count and substrate area data used to fit the regressions; Table S8, sensitivity analysis of characterization factor model parameters; Table S9, Monte Carlo uncertainty analysis of characterization factors (PDF)

SI2: Table S10, end point characterization factors [PDF*yr/m2] for seabed damage by activity and gear type per marine ecoregion (regional); Table S11, species-specific end point CFs [PDF*yr/kg] for bottom trawling for 51 commercially significant fish species (regional); Table S12, generic-fish end point CFs [PDF*yr/kg] for bottom trawling per marine ecoregion, including a globally aggregated area-weighted value (regional); Table S13, MER-level GEP-adjusted CFs [PDF*yr/m2], globally normalized; Table S14, species-specific GEP-adjusted CFs [PDF*yr/kg] for the 51 fish species; Table S15, generic-fish GEP-adjusted CFs [PDF*yr/kg] per marine ecoregion (XLSX)

■ AUTHOR INFORMATION

Corresponding Author

Sedona Anderson − *Industrial Ecology Programme, Department of Energy and Process Engineering, NTNU, Trondheim 7491, Norway;* orcid.org/0000-0002-4428-4290; Email: sedona.anderson@ntnu.no

Authors

Solveig Størdal Tveit − *Industrial Ecology Programme, Department of Energy and Process Engineering, NTNU, Trondheim 7491, Norway*

Konstantin Stadler − *Industrial Ecology Programme, Department of Energy and Process Engineering, NTNU, Trondheim 7491, Norway*

Francesca Verones − *Industrial Ecology Programme, Department of Energy and Process Engineering, NTNU, Trondheim 7491, Norway;* orcid.org/0000-0002-2908-328X

Complete contact information is available at: https://pubs.acs.org/10.1021/acs.est.6c10111

Notes

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