Probabilistic effect analysis - The Species Sensitivity Distribution

Probabilistic effect analysis-The Species Sensitivity Distribution

Type: Method

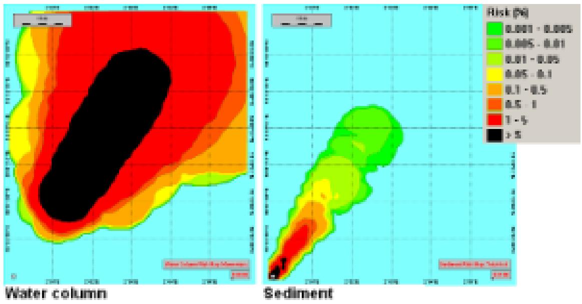
Project Phase: Planning and Design, Construction, Operation and Maintenance

Purpose: To assess the environmental impact of different interventions through species indicators

Requirements: Eco(toxico)logical knowledge of species and ecosystem at hand

Relevant Software: Statistical programmes (eg. R)

About



To assess the environmental impacts of human interventions, species indicators can be useful to evaluate the effects of different designs. The so-called Species Sensitivity Distribution (SSD) uses causal relationships between exposure level and effect level of individual species. By using exposure-effect data of multiple types of animals and plants, the sensitivity distribution of these species can be used to assess the risk of human interference more quantitatively. Although the SSD was originally developed to assess the ecological risk of toxicants, it also appeared to be applicable for non-toxic stressors. For instance, SSDs have been developed specifically for assessing the risk of suspended clay-particles and sedimentation for the environmental impact analysis of offshore oil and gas drilling activities. One advantage of having SSDs for non-toxic stressors is that they can be easily combined with those of toxic stressors, resulting in a single impact indicator - although such combinations remain to be fully validated.

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5 Basic steps towards Building with Nature

Related Building solutions

Feeder beaches

Managed realignment

Perched Beaches

Probabilistic effect analysis - The Species Sensitivity Distribution

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Related Tools

Cumulative Effect Assessment for Marine Environments

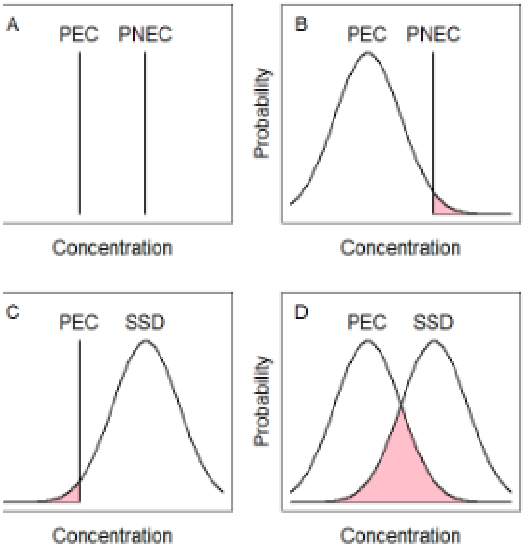
Including natural value in decision-making - Nature Index

Monte Carlo simulation and Social Cost Benefit Analysis

Probabilistic analysis of ecological effects - Cause-effect chain modeling

Probabilistic effect analysis - The Species Sensitivity Distribution

Species Response Curves for Seagrass



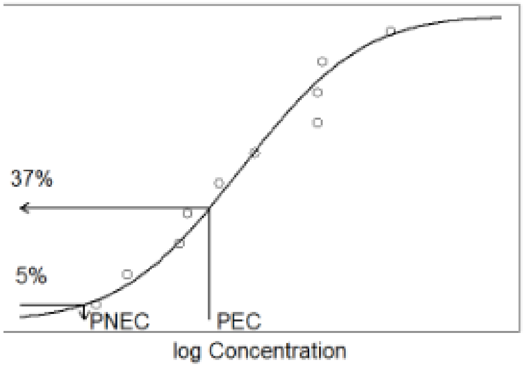
ssment.

Eco-toxicological risk assessment often makes use of Predicted Environmental Concentrations (PECs), i.e. the concentration that is expected after human
icted No Effect Concentrations (PNECs). PECs, are usually calculated with an appropriate model. The selection of such a model depends on the nature and source of the toxicant and the environmental system at hand. PNECs are usually based on lab studies and are often determined by applying an assessment factor to the most sensitive species (see box below). Traditionally, simply the ratio between PEC and PNEC is determined: a ratio larger than one indicates a potential ecological risk, whereas a value less than one indicates no risk . A disadvantage of this approach is that it doesn't quantify the risk: when the PEC:PNEC ratio is larger than one, we know that there is a potential risk but we don't know its magnitude. In other words, we expect that sensitive species are affected, but we don’t know which fraction of the community as a whole is at risk. Hence this approach is highly conservative, which can be a disadvantage.

There are four possible options for eco-toxicological risk assessment:

- Determine the ratio between the PEC and the PNEC.
- Take probability distribution of the PEC into account and determine the likelihood that the PEC exceeds the PNEC.
- Take the probability distribution of the species sensitivity into account and determine the fraction of species that is affected.
- Take the probability distributions of both the PEC and the species sensitivity into account.

Derivation of the PNEC
The PNEC is based on toxicological experiments with species in the laboratory, where the concentration-effect relationship for species is determined. PNECs are based on end-points (e.g., survival, growth, reproduction, etc.), which are usually measured in No Observable Effect Concentrations (NOECs) or the median lethal concentration (LC50). Traditionally, the PNEC is obtained by dividing the end-point of the most sensitive species tested in the lab by an assessment factor. This assessment factor is used to compensate for the uncertainty of lab to field, single to multi-species communities and other unknowns. This assessment factor is usually very large and depends on the amount and quality of data. It can range from 10 up to even 10.000 for marine species (Anonymous, 2003), where the highest factor is used when little information is available and smaller factors are used with increasing data availability. This approach focuses on information on the most sensitive species; information on other tested species is disregarded.



`tion.`

The SSD uses information on the sensitivity of all tested species (instead of only the most sensitive), by determining its statistical distribution. Of course, t

In an SSD, each marker represents the end-point of a toxicity test with a specific species, in other words: species sensitivity. The y-axis shows the cumulative probability distribution of species sensitivity, which can be interpreted as the PAF (37% in the presented example) at a certain exposure level. SSDs can be used in two ways: to derive 'safe' thresholds (PNECs), and to quantify the risk level at a specific PEC.

There are some important assumptions, on which both the SSD and the assessment factor approach rely:

- a sufficiently large number of species is used to construct the SSD;
- the selected species form a good representation of all species and ecological groups in the ecosystem;
- protecting individual species is good enough to protect the ecosystem. To strengthen this assumption, the use of exposure effect data of “ecosystem-relevant” or so called “keystone”

species is recommended.

Although there are some well-known critiques (e.g., Forbes and Forbes, 1993; Forbes et al., 2001), the SSD approach (for toxicants) has been validated by several studies (e.g., Hose and Van den Brink, 2004; Selck et al., 2002; Van Wijngaarden et al., 2005). As indicated above, SSDs have originally been developed for risk assessment of toxicants. Only in recent years has the method been implemented to assess risk from non-toxic stressors. For toxic stressors laboratory test protocols are well defined and standardised, which reduces uncertainty in the risk calculations with SSDs. For non-toxic stressors, such standardisation is often not in place. This must lead to a higher uncertainty in the SSD risk calculations, but the extent to which this is the case remains to be investigated.

The SSD has shown to be a practical and useful tool in environmental risk assessment. It is a relatively straightforward approach which can report environmental impact by a single indicator and potentially allows integration of risks from multiple stressors. These properties are advantageous in environmental risk management.

Building with Nature interest

In BwN-projects, SSD can be used in various ways. In adaptive management, for instance, SSDs can be used to continuously evaluate the risk of human interference based on in situ data. SSDs can also be used to estimate the effects of different alternatives in advance (e.g. BwN concepts versus a traditional approach), via exposure effect relations for relevant stressors. For the latter, they have to be used in combination with tools that estimate environmental exposure levels (PECs). SSDs are therefore relevant for BwN phases ranging from design to (adaptive) construction management.

How to Use

SSDs can be used by eco(toxico)logists with sufficient knowledge of the ecosystem and its species. The application of SSDs is one form of probabilistic effect analysis (PEA). Please refer to the tool Probabilistic analysis of ecological effects - Cause-effect chain modeling, for a description of another form of PEA.

1. Applicability of SSDs for environmental impact assessments
2. Use of the environmental impact factor
3. Spatial and temporal issues
4. Conclusion and recommendation

>> Read more

1. Appliccability of SSDs for environmental impact assessments

The first steps are to determine whether SSDs may be of use to evaluate the effects of human interventions:

- **Step 1:** select the most relevant ecosystem characteristics that are expected to be impacted by the planned human intervention (make use of results from other tools see: Systems analysis);

- **Step 2:** try to determine (e.g. by expert judgment) if and how this intervention may affect the species present.

- **Step 3:** if a significant effect is expected or cannot be excluded, determine whether SSDs can be of use to assess these effects (depending on the availability of data, time, budget, etc.).

The next steps apply when SSDs are expected to be useful. There is no readily available approach for applying SSDs to a project, but some general steps can be followed. For offshore drilling activities an approach - the Environmental Impact factor (EIF) - has been developed (see practical applications). This may serve serve as an example of how SSDs can be used:

- **Step 4:** determine the most important stressors for the species present, based on knowledge of the ecosystem and the impact of the specific interference on the ecosystem.

- **Step 5:** make a good (large and representative) selection of species from the ecosystem for which exposure effect data for the selected stressors is available or can be derived (e.g. by lab experiments).

- **Step 6:** carry out the impact analysis. See for an example the Environmental Impact Factor (EIF) in the practical application.

2. Use of the Environmental Impact Factor

As already mentioned, the Environmental Impact factor (EIF) approach has been developed for offshore drilling activities. The drilling discharge model (EIFDD) runs in the Marine Environmental Modelling Workbench (MEMW), developed by SINTEF, Norway. Practically anyone with the proper (few days) training can operate this model. SINTEF’s office in Trondheim can be contacted for inquiries on the model. Keep in mind, however, that this implementation focuses on drilling discharges and produces a generic impact assessment. As indicated before, the model might be adapted to marine construction activities, like dredging.

As building with nature is highly site-specific, it is not desirable to use a generic approach. Therefore, the EIFDD cannot be applied as is to marine construction activities. Rather should SSDs be developed for specific situations. Such developments will require more specialised (ecological) knowledge, as developed under the Environmental Risk Management System (ERMS) program, for instance.

As a case study one may want developing an SSD for tropical corals. In that case (preferably consistent) effect data on tropical corals needs to be collected or may already be available from other tasks within Building with Nature. Once SSDs have been developed, they should be combined with predicted or measured turbidity levels.

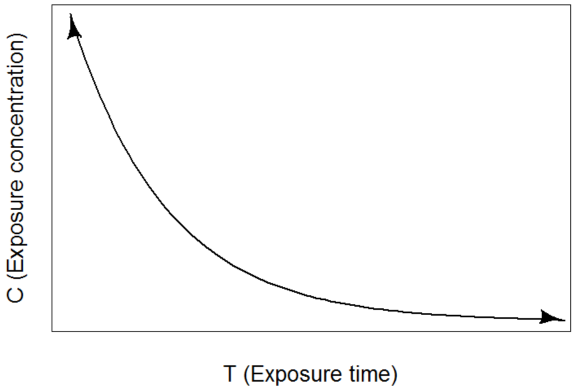
3. Spatial and temporal issues

When the SSD approach is going to be used for marine construction activities, some issues with respect to spatial and temporal aspects need to be considered. Usually SSDs describe the ecosystem sensitivity in a generic manner. They don’t have a spatial dimension, for instance. This will work fine if the species are distributed homogenously in space, but if this is not the case and the ecosystem sensitivity to stressors varies in space, it may be necessary to develop more than one SSD, each representing a specific region. Of course, such a more specific approach requires more location-specific data.

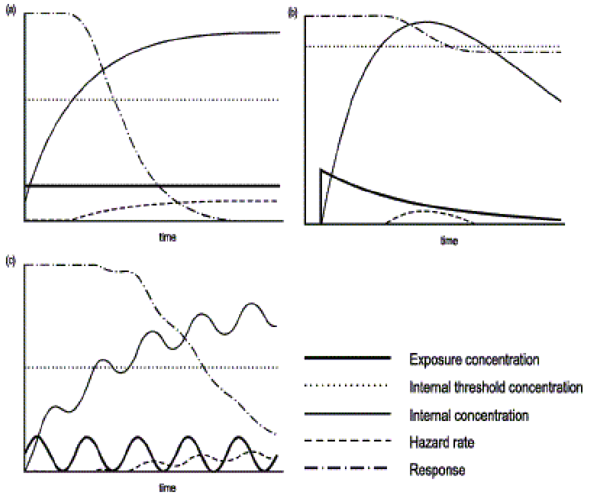
For adaptive construction management this may not be sufficient, as operations are time-dependent and operational decisions need to be made at short notice. Another issue to be considered is therefore the factor 'time'. First of all, in toxicology, SSDs are based on chronic end-points, which is actually a worst-case approach. The idea behind this is that if a species is not affected after chronic exposure, it will certainly not be affected after a shorter exposure. Unfortunately, examples of time-dependent SSDs are scarce in literature. In fact, there is only one publication (Smit et al., 2008c) in which a time-dependent SSD (for hydrogen peroxide exposure) is actually developed. In this approach several constant concentrations were tested, and effects on several species were recorded as a function of time. This gives a simple and straightforward relationshio between exposure time and effect, known as Habers' Law (Karman, 2000). To achieve the same effect with higher exposure concentrations, a shorter exposure duration is required and vice versa. Note that this relationship applies to constant stress levels and is not suitable for fluctuating exposure levels.

For toxicants models are available that can describe effects of fluctuating exposure concentrations. For example some models consider the kinetics (uptake and excretion speeds) of toxicants. With such kinetic information internal concentration in a species can be calculated from a (fluctuating) external concentration. This internal concentration can then be compared with internal effect threshold levels, in order to determine effects as a function of time (see for example Figure, Graphical representation of inverse hazard model, applied to three hypothetical cases, ref Karman, 2000). A similar, more mechanistic approach (e.g., considering uptake, storage and excretion processes and target sites in the organism) would also be possible for non-toxic stressors (such as suspended matter), although this requires knowledge on the mechanism through which these stressors affect the organism, as well as data to quantify them.

Although the latter approach (fig 4c) accounts for the factor 'time' in a more realistic and sophisticated way, it doesn't take the recovery potential of a species into account. Furthermore, it is difficult to implement this approach in a simple SSD approach, as effects will not only depend on the exposure time and the current exposure level, but also on the exposure history.



As these examples illustrate, including spatial and temporal effects complicate the SSD methodology. Therefore, it is advisable to start with a simple approach, viz. a generic SSD based on worst-case end-points (e.g. chronic NOECs) and to add data and realism (hence complexity) only when required. This is actually the approach implemented by the Norwegian offshore oil and gas industry as described under Practical Applications section.



Graphical representation of inverse hazard model

4. Conclusions and recommendations

SSDs can be applied:

- to identify and rank the substances by their contribution to the overall probability of affecting sensitive species;
- to select, on an environmental cost/benefit basis, (potential) discharge reducing measures (BAT /BEP techniques, see Practical Applications section for more details);
- to compare the environmental risks of different projects/operational strategies;
- to serve as basis for environmental monitoring.
  A comparison with effect data for other types of suspended particulate matter should indicate the general applicability of the derived SSDs for activities other than offshore oil and gas drilling. The fate of the sediment particles during and after marine construction activities should be modeled. Available exposure models should be identified.

Practical Applications

The SSD approach has been implemented by the offshore oil and gas industry to evaluate the environmental impact of drilling discharge scenarios. Such discharge plumes are to some extent comparable with dredging plumes. Therefore, the approach may also be applicable to evaluate the effects of dredging. The implementation of SSDs in the EIF by the oil and gas industry is illustrated in this section.

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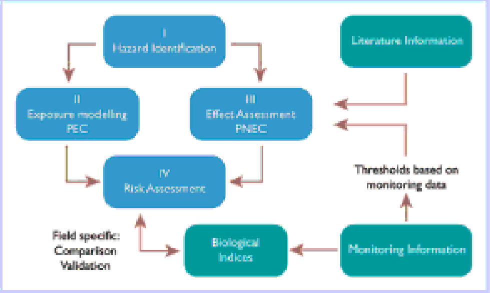


`Framework for the environmental impact`

`factor for drilling discharges (EIF DD).`

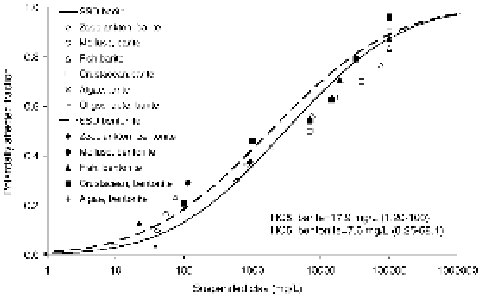
The environmental impact factor for produced water (EIF PW) was developed in the year 2000 by the oil and gas exploration and production companies active on the Norwegian shelf, as a risk assessment tool for environmental management of produced water discharges. The EIF PW is an indicator of environmental risk, with the purpose to aid the industry in developing a 'zero harm' strategy and selecting cost-benefit based solutions. The Norwegian authorities presently require using this tool in reporting and planning of management actions to reduce potentially harmful environmental effects associated with produced water discharges (Singaas et al., 2008). In order to further develop the toolbox for environmental management, the Environmental Risk Management System (ERMS) Joint Industry Project was established to develop an EIF for drilling discharges (EIF DD). T he framework for the EIF DD indicates the different steps in the risk assessment process (Singaas et al., 2008) (see figure). Six stressors related to the discharge of drilling waste to the marine environment were identified (see step I in figure and table below), of which two occur in the water column (toxicity of chemical substances and physical effects of suspended clay particles) and four in the sediment (toxicity of chemical substances, burial of organisms, oxygen depletion, and change in sediment structure).

Table 1: Stressors of drilling and dredging operations

Environmental compartment	**Stressor (Drilling operations)**	**Stressor (Dredging operations)**
Water Column	Toxicity of chemical substances	Toxicity can be relevant if dredged material is contaminated
	Physical effects of suspended clay particles	Physical effects of suspended sediment
Sediment	Toxicity of chemical substances	Toxicity is relevant when dredged material is contaminated
	Burial of organisms	Burial of organisms
	Oxygen depletion	Oxygen depletion
	Change in sediment structure	Change in sediment structure

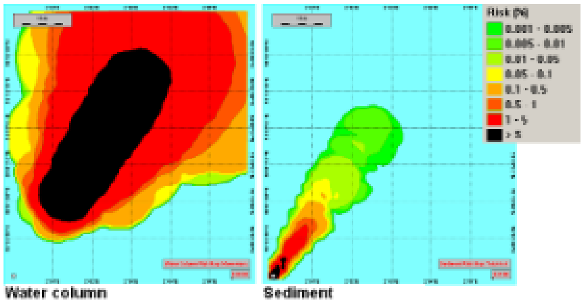
To assess the exposure (step II) the fate of each of the stressors is modelled. The effect assessment (step III) is based on laboratory tests and field measurements reported in literature. For each stressor identified a species sensitivity distribution (SSD) was constructed in such a way that the 5th percentile of the distribution corresponds to the Predicted No Effect Concentration (PNEC) of the constructed species. For risk assessment (step IV) for single stressors, the ratio of predicted environmental concentration (PEC) and PNEC is prescribed by the EU Technical Guidance Document on risk assessment (Anonymous, 2003). Sensitivity distributions for the various stressors were used to calculate the potentially affected fraction of species as a result of the simultaneous exposure to multiple stressors (msPAF). Monitoring information was available for validation purposes. Generally, a good correlation was found between model results and measured data (Singaas et al., 2008).

For suspended clays the data collection resulted in a database with effect concentrations for marine species, distinguishing different effect types, such as survival, feeding behaviour, growth, mobility, reproduction, oxygen consumption, and effects on the gastrointestinal tract (Smit et al., 2008a). Data were obtained from exposure studies with clay-sized particles (i.e. attapulgite, bentonite, clays, and barite) and also with various types of water-based drilling fluids. Most data were available for the species groups phytoplankton, zooplankton, crustaceans (excluding zooplankton), molluscs, and fish.



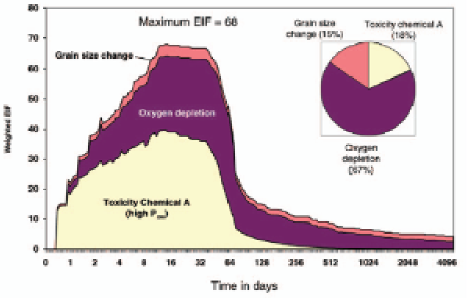
SSDs and the corresponding hazardous concentration for 5% of the species.

The SSDs, based on collected 50% effect concentration (EC50) values for mortality resulting from suspended barite and bentonite exposure, present median values and 5-95% confidence intervals (see SSDs with corresponding hazardous concentration) (Smit et al., 2008a). Filter feeding zooplankton and molluscs are, together with algae, among the sensitive taxonomic groups, while benthic crustacean and siphon feeding molluscs are relatively insensitive (Smit et al., 2008a). This supports the hypothesis that organisms living in the benthic boundary layer of fine sediments are accommodated to deal with elevated turbidity and sedimentation.



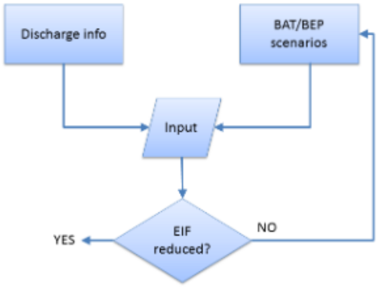
Example of spatial risk map for water column (left) and sediment (right).

The value of the EIF is related to the recipient water volume and sediment surface area where msPAF exceeds 5% (Smit et al., 2008b) (se spatial risk map). The selected unit for the EIF for the 2 compartments is a water volume of in units of 100 m x 100 m x 10 m for the water column and 100 m x 100 m surface area for the sediment. An advantage of the EIF method is that it calculates the overall risk and the contribution to it from each stressor (see output of DREAM).



Typical EIF DD output of the Dose-Related Risk and Effect Assessment Model (DREAM).

The EIF in its present form should be regarded as a management tool to identify and rank the most harmful discharges (and components of those discharges). This enables operators of production fields to evaluate mitigating measures on a cost/benefit basis, when applying the Best Available Technology (BAT) for treatment of produced water of all production fields (Smit et al., 2003). The EIF provides the possibility to assess different management options, enabling the field operator to prioritise the implementation of BAT for the specific field and to document Best Environmental Practice (BEP) (see EIF application for management option).



Application of the EIF in relation to management options (Smit et al., 2003).

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>> Read more

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