



## Exploring methods for predicting multiple pressures on ecosystem recovery: A case study on marine eutrophication and fisheries

Uusitalo, Laura; Korpinen, Samuli; Andersen, Jesper H.; Niiranen, Susa; Valanko, Sebastian; Heiskanen, Anna Stiina; Dickey-Collas, Mark

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## Research papers

## Exploring methods for predicting multiple pressures on ecosystem recovery: A case study on marine eutrophication and fisheries

Laura Uusitalo<sup>a,\*</sup>, Samuli Korpinen<sup>a</sup>, Jesper H. Andersen<sup>b</sup>, Susa Niiranen<sup>c</sup>, Sebastian Valanko<sup>d</sup>, Anna-Stiina Heiskanen<sup>a</sup>, Mark Dickey-Collas<sup>e</sup>

<sup>a</sup> Marine Research Centre, Finnish Environment Institute, Finland<sup>b</sup> NIVA Denmark Water Research, Denmark<sup>c</sup> Stockholm Resilience Centre, University of Stockholm, Sweden<sup>d</sup> International Council for the Exploration of the Sea, Denmark<sup>e</sup> International Council for the Exploration of the Sea, DTU Aqua National Institute of Aquatic Resources, Technical University of Denmark, Denmark

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## ABSTRACT

Efforts to attain good environmental status in the marine realm require decisions which cannot be done without knowledge of effects of different management measures. Given the wide diversity of marine ecosystems, multitude of pressures affecting it and the still poor understanding on linkages between those, there are likely no models available to give all the required answers. Hence, several separate approaches can be used in parallel to give support for management measures. We tested three completely different methods – a spatial impact index, a food web model and a Bayesian expert method. We found that a large uncertainty existed regarding the ecosystem response to the management scenarios, and that the three different modelling approaches complemented each other. The models indicated that in order to reach an improved overall state of the ecosystem nutrient reductions are the more effective of the two management variables explored, and that cumulative effects of the management of nutrient inputs and fishing mortality are likely to exist.

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## 1. Introduction

Making informed decisions to achieve cost-effective improvements in environmental status requires knowledge about the ecosystem responses to the changes in managed pressures caused by human activity (Borja et al., 2010; Borja, 2014; Duarte et al., 2015). Marine science has collated a large body of evidence of impacts of various human activities to the marine ecosystems; however this information will by its very nature always be considered incomplete (Knowlton and Jackson, 2008; Borja, 2014). Moreover, the vast majority of this knowledge comes from marine environments where human pressures have increased; and if we wish to assess the recovery rate of the ecosystem as the pressures are relieved, the possibility of hysteresis in the recovery process should be recognised (Duarte et al., 2015). The issue is further complicated by “shifting baselines”, i.e. the gradual change in variables such as climate, atmospheric pollution, patterns of human use, etc. (Duarte et al., 2009). The challenge in designing the optimal management strategy is two-fold: we need to assess the

likely recovery paths of the ecosystem considering likely reductions in pressures, and we must understand the cumulative, or synergistic, effects of these processes during recovery (Borja, 2014).

When acting simultaneously, pressures may have effects that are additive, i.e., the combined effect can be evaluated by simply adding up the individual effects of the pressures; but often they have cumulative, i.e. synergistic or antagonistic effects, either strengthening or weakening each other (Griffith et al., 2011, 2012). Understanding these effects is needed in order to help the manager select and implement an effective set of measures to protect the ecosystem, and to predict ecosystem recovery when these pressures are relaxed. There are many cases where the deterioration of the ecosystems has been experienced and documented (Myers et al., 1997; Möllmann et al., 2009), but less cases where there are evidence of pressure relief and subsequent improvement of the environmental status (however see Carstensen et al., 2006; Andersen et al., 2015b; Riemann et al., 2015). Therefore, the current understanding in modelling and prediction of ecosystem recovery is not sufficient to provide operational management tools for quantitative decision-making in situations where multiple pressures are impacting the environment (Francis et al., 2011; Planque, 2015). Managers facing this fundamental uncertainty in

\* Corresponding author.

E-mail addresses: [laura.uusitalo@ymparisto.fi](mailto:laura.uusitalo@ymparisto.fi), [laura.uusitalo@iki.fi](mailto:laura.uusitalo@iki.fi) (L. Uusitalo).

knowledge and modelling tools have to make decisions and proceed in managing human activities, using the best available scientific information. Therefore the understanding of the trade-offs and potential synergies between various actions and their effects on the marine environment is crucial (Lester et al., 2013). Managers need advice on how the effects of the management measures propagate beyond their primary target (Samhoury et al., 2011), an example being how the nutrient loading reductions in a eutrophied system have consequences beyond phytoplankton biomass to food web structure, the benthos, etc. Further, they need the best available estimates about the interactions of various management measures; whether they are likely to give boost to each other (i.e. be synergistic) or dampen each other's effects (antagonistic), or whether one of them only works if the other is implemented at the same time (Judd et al., 2015).

The need to manage human activities and predict the outcome in the environment has increased with the environmental legislation (e.g. Marine Strategy Framework Directive (MSFD), European Union, 2008), emerging maritime spatial planning (European Union, 2014) and the increased awareness of impacts of multiple human activities on marine ecosystems (Korpinen et al., 2012; Halpern and Fujita, 2013; Korpinen et al., 2013). The MSFD requires EU Member States to create and regionally coordinate programmes of management measures to reach good environmental status (GES) of Europe's seas. The challenge of this requirement is underlined by the fact that only part of the pressures are measured quantitatively. Likewise, the impacts of some pressures are not very well understood, and building quantitative

models is a challenge (as compared to, e.g., the impact of a fisheries on a well monitored fish stock). Further, for example marine biodiversity has been divided into categories, which are often too broad to be used directly in models that aim to estimate potential effects of management measures. For example, marine ecosystem complexity is often divided into three broad categories: (1) species abundance and condition, (2) quality of habitats and their communities, and (3) food web structure (European Union, 2008, 2010; HELCOM, 2010). Ecosystem assessments in the Baltic Sea and NE Atlantic are recent examples of this approach (HELCOM, 2010; OSPAR, 2010). Due to the difficulty in capturing the processes of an entire ecosystem and pressures affecting them, ecosystem models and assessments have used indicator species (e.g. keystone species, predominant food web elements) which simplify the multitude of interactions and reflect broad-scale phenomena in the system (Heslenfeld and Enserink, 2008; HELCOM, 2010; OSPAR, 2010; ICES, 2015a).

The aim of this study is to explore different approaches to estimate the potential outcome of pressure reductions by including two well-known, and in the study area, central, anthropogenic pressures – nutrient inputs and fishing – with different reduction scenarios (alone and together). We approached this challenge using three types of approaches: (1) a spatial model for cumulative impacts (additive approach), (2) a food web model, and (3) a Bayesian model harnessing expert knowledge. We present the approaches and results and discuss their pros and cons in a challenging management situation.



Fig. 1. The Baltic Sea.

## 2. Methods

### 2.1. Study area: Baltic Sea

This study was carried out in the Baltic Sea, a brackish water body bounded by the Scandinavian Peninsula, the mainland of Europe, and the Danish islands (Fig. 1). It is connected to the North Sea by the Danish Straits and Kattegat. Being relatively shallow (average depth 52 m), almost enclosed by land, and with a drainage basin approximately 4 times the area of the sea (Leppäranta and Myrberg, 2009), the marine ecosystem of the Baltic Sea is particularly susceptible to human activities and pressures, e.g. fishing, pollution (of both nutrient and contaminants), physical modification, introduction of non-native species as well as climate change (Korpinen et al., 2012; Andersen et al., 2015b). The shallow and narrow straits at its outlet give long water residence times, which together with basin topography and eutrophication have resulted in wide-spread hypoxia of deep waters (Zillén et al., 2008; Carstensen et al., 2014). In addition, due to the brackish water, the species richness is low and declines towards the less saline sub-basins in North and East, as some species are restricted by low salinity, others by high salinity or an inappropriate temperature regime (Ojaveer et al., 1981; Lehtonen and Rask, 2004; Ojaveer et al., 2010). Consequently, many of the Baltic species live at the edge of their environmental tolerance limits, and hence the biological system may have reduced resilience towards external perturbation (Myers, 1998, 2001). Due to intensive fishing pressure, the dense human populations and high levels of industrialisation and agriculture, the ecosystem health is significantly impaired (HELCOM, 2010).

### 2.2. Selection of the variables: fishing and nutrient inputs in the Baltic Sea

This pilot study focuses on nutrient inputs and fishing in the Baltic Sea. The objective is to address pressures of which previous knowledge and data, as well as models, are available. The pressure–impact relationships studied are reduced into a manageable number that can be tested concurrently.

Over 90% of the Baltic Sea commercial catches in weight come from three fish species: cod (*Gadus morhua*), herring (*Clupea harengus membras*), and sprat (*Sprattus sprattus*). Flatfish, mainly flounder (*Platichthys flesus*), comprise a small minority of total catch (Zeller et al., 2011). Other species (such as salmon (*Salmo salar*), brown trout (*Salmo trutta*), eel (*Anguilla anguilla*)) are highly prized but their catches are low. Further, there is small-scale professional fishing of several coastal species such as northern pike (*Esox lucius*), pikeperch (*Sander lucioperca*), perch (*Perca fluviatilis*) and whitefish (*Coregonus lavaretus*). Recreational fisheries exist along the coastal regions but are not considered in this study.

The effects of nutrient inputs on the marine ecosystem are very well known and described in several review publications and meta-analyses (e.g. HELCOM, 2009; Fleming-Lehtinen et al., 2015). Direct and indirect effects of nutrient inputs from land, air and sea-based sources lead to eutrophication, which has become partly self-sustaining in the Baltic Sea due to internal loading, i.e., nutrients being released from the sediments under hypoxic conditions (Vahtera et al., 2007). Consequently, eutrophication has been identified as the worst environmental problem for the Baltic Sea. Its main effects in the coastal zone include increased growth of filamentous algae and consequent suppression of macroalgae and vascular plants, which affects the habitat structure, ecosystem biodiversity and food web dynamics. In the pelagic habitat, eutrophication leads to increased growth of planktonic algae and harmful algal blooms, causing increased light attenuation and reduced visibility. This also changes the habitat quality of pelagic

environment and food web structure, and increases the sedimentation of organic matter, leading to increased oxygen consumption and anoxia in the benthic system (e.g. Elmgren, 2001; Laine, 2003; HELCOM, 2009; Suikkanen et al., 2013; Carstensen et al., 2014).

### 2.3. Models and scenarios

In order to address potential recovery of the ecosystem state when pressures are reduced, we developed a simple scenario-based framework. This builds on three methods: (1) spatial visualisation of GES and cumulative pressures and impacts (Korpinen et al., 2012), (2) the Central Baltic Sea food web model (Tomczak et al., 2012; Niiranen et al., 2013) to predict changes in fish biomass, and (3) a Bayesian network model encoding expert opinion under the reduction of nutrient inputs and fishing (see e.g. Uusitalo, 2007).

The models differ from each other in their modelling technique and assumptions, and they take their input and produce their results in different forms and resolutions. For example, in the food web model the effect of nutrient loads is incorporated via changes in primary productivity. Thus, nutrient load scenarios cannot be used directly, but biogeochemical models are needed to link them with relevant oceanographic and biogeochemical processes. This makes it impossible to come up with scenarios that are exactly similar, or produce results that would be perfectly comparable with each other. We argue that this drawback is an acceptable price to pay for the benefit of gaining insight from three very different approaches, however; where the different models agree, a high confidence on the reliability of the results can be assumed, whereas differences between the models show that the knowledge base behind the models may need to be revised.

We set up a series of scenarios for the reduction of nutrient inputs and fishing mortality. The basis of these scenarios was to compare (1) business-as-usual scenario, in which current or recent nutrient loading and fishing mortality levels are maintained but no further restrictions are implemented; (2) a 30% cut in the pressures (nutrient inputs and fishing mortality), and (3) 60% cuts in the pressures. See Table 1 for a summary of the definitions of the scenarios in each case.

#### 2.3.1. Approach 1 – spatial models

Spatial models quantifying human pressures and impacts have been developed in several marine regions (e.g. Ban et al., 2010; Coll et al., 2012; Korpinen et al., 2012; Micheli et al., 2013) and globally (Halpern et al., 2008). The mapping of cumulative impacts in the Baltic Sea was developed and applied in the HELCOM Initial Holistic Assessment of ecosystem health of the Baltic Sea (HELCOM, 2010). Despite multiple assumptions in the methodology (Halpern and Fujita, 2013) as well as an incomplete understanding of synergistic, antagonistic and cumulative effects of human activities, such models are as far as we know the only operational attempts to map cumulative impacts. These models assume that the pressures from the activities are additive. In this study, we used the HELCOM impact assessment, covering data from 2003 to 2007, and modified this model by including the nine management scenarios. We assumed spatially evenly distributed reductions over the case study area and estimated cumulative impacts ( $I$ ) for a 5 km × 5 km grid using the formula

$$I = \sum_{i=1}^n \sum_{j=1}^m P_i \times E_j \times \mu_{i,j}$$

where  $p_i$  is the log-transformed and normalised value of an anthropogenic pressure (scaled between 0 and 1) in an assessment unit,  $E_j$  is the presence or absence of an ecosystem component  $j$  (1

**Table 1**  
The pressure scenarios implemented in the three evaluated models.

Pressure	Scenarios implemented in the spatial model	Scenarios implemented in the food web model	Scenarios implemented in the expert judgement model
Fisheries mortality	Reduction (0, 30, or 60%) to the fishery landings of all landed species (mainly herring, sprat, and cod) in all areas compared to 2003–2007 level.	Reduction (0, 30, or 60%) to the fishing mortality of Central Baltic Sea herring, sprat, and cod compared to 2004–2006 level.	Reduction (0, 30, or 60%) to the fishing mortality of herring, sprat, and cod in all basins compared to 2014 level.
Nutrient input	Reduction (0, 30, or 60%) to the nutrient inputs in all areas compared to 2003–2007 level.	Reference: nutrient loads equal the average loads for 1995–2002; Baltic Sea Action Plan (BSAP): approx. 25% reduction of N and 60% reduction of P from the reference period 1997–2003 in land based nutrient sources within the Baltic Proper catchment area, combined with a 50% reduction in atmospheric nitrogen deposition and BSAP nutrient load reductions in other catchments (Gustafsson et al., 2011).	Reduction (0, 30, or 60%) of the limiting nutrient (phosphorus and/or nitrogen) loading from all sources (point and diffuse, all countries), compared to 2014 level.

or 0, respectively), and  $\mu_{ij}$  is the weight score for  $p_i$  in  $E_j$  (range 0–4) (Halpern et al., 2008; Korpinen et al., 2012). In brief, the pressure intensity was estimated by the underlying activities in the grid cells, such as number of wind turbines, biomass of caught fish, average number of ships or amount of nitrogen deposited from atmosphere (see Korpinen et al., 2012). The ecosystem components consisted of underwater habitat maps, water-column habitat maps, distribution areas of marine mammals, and spawning and nursery areas of cod, each of which was modelled as either present (value=1) or absent (value=0) in an assessment unit. The weight scores were formed on the basis of three criteria – functional impact, recovery time and resistance of the ecosystem against the pressure – by an expert panel through a workshop and a following expert survey. In total, there were 52 GIS data layers depicting human-induced pressures and 14 layers depicting species and habitat distribution for the years 2003–2007. Values of each  $P$  were multiplied by each  $E$  and their common  $\mu$ . If a pressure or an ecosystem component did not occur in an assessment unit, or the  $\mu$  value was zero, this element got value of zero and did not therefore contribute to the sum that is the BSII score. Detailed description of the pressures, ecosystem components, weighting scores and the calculation of the index were given by Korpinen et al. (2012) and the method has been further discussed by Halpern and Fujita (2013).

In the spatial model, we created the alternative scenarios by calculating the impact index using reduced fishing and nutrient inputs (see scenarios in Table 1). Second, we groundtruthed the original cumulative impact result by comparing it with the latest biodiversity assessment of the Baltic Sea (HELCOM, 2010). The HELCOM assessment was made by the assessment tool BEAT (Andersen et al., 2014), which makes use of quantitative indicators and associated thresholds for GES. Each of the indicators has a pre-defined GES threshold, as agreed in HELCOM (<http://www.helcom.fi/baltic-sea-trends/biodiversity/indicators/>), and the BEAT tool integrates the biodiversity status according to those thresholds. Thus, each assessed area can be defined to GES and given a quantitative distance from that threshold. In that assessment, the integrated biodiversity status was estimated for nine sub-basins as a number below or above 1.0 which reflected the GES (Andersen et al., 2015b; Fleming-Lehtinen et al., 2015). The comparison of the two assessments showed a significant negative correlation between cumulative impacts and the status of biodiversity (Pearson's  $r = -0.70$ ;  $p = 0.034$ ;  $n = 9$ ) and also gave an indicative threshold value of 100 for cumulative impacts causing disturbed environmental status (Andersen et al., 2015b). As we used in this study the same data set, we were able to use this value as a possible threshold above which cumulative impacts drive the system to a disturbed state of biodiversity. We note however, that the threshold value 100 is not a fixed

value but an average of potential values in a linear correlation line and it varies according to input data. Setting the threshold to the map of cumulative impacts, one can indirectly estimate the area which is in good environmental status, but such an indirect estimate does not take account of time lags in pressure impacts (in case of new pressures) or ecosystem recovery (after reduction of pressures). By using the same threshold in the pressure reduction scenarios, we estimated the increase of area in good environmental status in the case study area.

### 2.3.2. Approach 2 – food web model

The Central Baltic Sea Ecopath with Ecosim (EwE, Christensen and Pauly, 1992) food web model BaltProWeb (Tomczak et al., 2012) can be used to project future changes in population abundances on different trophic levels (e.g. Niiranen et al., 2013 and references therein). Recently, the model was run with a combination of different climate, nutrient load and fishing scenarios, while being off-line coupled with an ensemble of three biogeochemical Baltic Sea models (Meier et al., 2012a; Niiranen et al., 2013). In this study, the BaltProWeb model was forced with output from the biogeochemical model RCO-SCOB1 (Meier et al., 2003; Eilola et al., 2009). Regional biogeochemical model output from two nutrient load scenarios were implemented for 2007–2039 in the food web model: reference (REF; average nutrient loads for reference period 1995–2002) and reductions according to the Baltic Sea Action Plan (BSAP, HELCOM, 2007) (Table 1). Both scenarios are described in detail in Gustafsson et al. (2011). Model was calibrated with data for 1974–2006, and hence the fishing reduction scenarios are implemented stepwise from 2007 onwards such that the scenario-specific values for  $F$  are reached by 2015. The assessment  $F$  values were not used for 2007–2014 in the model runs due to present uncertainties in current stock estimates of Eastern Baltic cod. Differing from other modelling approaches applied in this study an intermediate future climate change scenario ECHAM5-r1-A1B (Nakićenović, 2000; Niiranen et al., 2013) was assumed in combination to every nutrient load-fishing scenario. Adding a climate scenario affects the biogeochemical model projections of nutrient concentrations in the sea, due to changes in precipitation in the catchment area, oceanographical properties and decomposition rate of sediment organic material (Meier et al., 2012b). Changes in climate may also affect species interactions. The model results of Niiranen et al. (2013), for example, show that projected decreases in salinity and oxygen may have a negative effect on cod production. Sprat, on the other hand, may benefit from projected increases in sea-surface temperature. However, most climate change scenario effects were detected in the longer-term (2050–2098) food web model projections (results not shown).

Furthermore, mean trophic level of catch (MTL), total system throughput (TST), and ascendancy were evaluated for each of the scenarios. These ecosystem indicators can show changes in the food web structure or function, and have previously been applied by Tomczak et al. (2013) to address past ecosystem change in the Central Baltic Sea. MTL describes changes in the trophic level of total fisheries catch, reflecting both changes in the catch composition and potential changes in the diet of commercial fish (Pauly et al., 1998). TST describes the sum of all food web flows modelled including outward flows, such as catch, respiration and other export (Finn, 1976). An increase in TST denotes increase in production or turnover rate of the system, and can hence indicate of system growth. Relative ascendancy is the index of food web organisation and development, i.e., the channelling of flows, of the system (Ulanowicz, 2001). Increase in ascendancy can indicate of the polarisation of flows within the food web, and thus may decrease system resilience to external disturbance. How these indicators are calculated in EwE models is described in detail by (Heymans et al., 2007, 2014).

### 2.3.3. Approach 3 – expert judgement

Management options that cover a wide range of environmental factors, dynamic processes and interactions, geographical areas, and varying temporal scales are rarely quantitatively evaluated by one, coordinated research project or model (Borsuk et al., 2004; Barton et al., 2012). In these cases, expert judgement can be used to help populate the decision support models, especially when the available evidence is limited, of mixed quality, or only indirectly relevant (O'Hagan, 2012). Experts' judgements are invaluable in many applications due to their ability to assimilate complex and equivocal evidence, interpret it in the light of broader experiences. Expert assessment has, therefore, been established as a methodology for obtaining estimates of relationships that cannot be or are too expensive or impractical to observe directly, such as

hypothetical scenarios (e.g. Krueger et al., 2012 and references therein). As the current case study aims to span over the whole ecosystem, including both open-water and coastal habitats and all ecosystem components, no models were available that are able to simulate the whole range of ecosystem from bio-geo-chemical processes to top predators, and various coastal habitats to open water, in all the Baltic basins. Therefore, in order to get a compilation of the current best understanding of the whole-system responses to the selected management scenarios, we organised a workshop where 14 experts of marine ecology and fisheries discussed and gave their assessment of the potential effect of the selected management scenarios on the overall ecosystem status of the Baltic Sea.

The expert elicitation was executed in two rounds: First, the experts were given the brief definitions of the scenarios: Business-as-usual, 30% reduction in fishing/nutrient loading, 60% reduction in fishing/nutrient loading. They were asked to provide a probability distribution for each 9 scenarios (all combinations of the 3 scenarios per measure) on the following scale: '2014 state', 'slightly better', 'slightly worse', 'much better', and 'much worse' by filling in a simple table (Fig. 2). During this first round, the experts were able to familiarise themselves with the assessment scheme, and they noticed the need to define the variables in more detail.

The first assessment round was then followed by discussions in which the experts agreed on the exact definitions of the aspects to be evaluated (Table 2). Thereafter, the different scenarios were discussed in two parallel sub-groups, and finally, each expert gave their estimates of the probabilities again, bearing in mind the definitions of the variables and taking with them anything they found useful in the discussions.

A Bayesian network model (BN) was constructed from the expert assessments using the GeNIe software (Decision Systems Laboratory of the University of Pittsburgh, <http://genie.sis.pitt.edu/>). The model structure consisted of three nodes: fishery

		Fisheries management policy					
		0		-30		-60	
Eutrophication management policy	0	much better		much better		much better	
		slightly better		slightly better		slightly better	
		2014 state		2014 state		2014 state	
	slightly worse		slightly worse		slightly worse		
	much worse		much worse		much worse		
		0 %		0 %		0 %	
	-30	much better		much better		much better	
		slightly better		slightly better		slightly better	
		2014 state		2014 state		2014 state	
slightly worse		slightly worse		slightly worse			
much worse		much worse		much worse			
	0 %		0 %		0 %		
-60	much better		much better		much better		
	slightly better		slightly better		slightly better		
	2014 state		2014 state		2014 state		
slightly worse		slightly worse		slightly worse			
much worse		much worse		much worse			
	0 %		0 %		0 %		

Fig. 2. The questionnaire form that was used to elicit the experts' opinions about the ecosystem state given each combination of the management policies.

**Table 2**  
The definitions of the variables in the expert assessment exercise as agreed by the expert panel.

Model aspect	Definition used
Spatial scale	Baltic-wide: both coast and offshore
Ecosystem status	The balance and functioning of the benthic-pelagic food chain of the Baltic Sea basins. Compared to the current (year 2014) status, with 5 alternative classes: much worse, slightly worse, approximately similar, slightly better, much better.
Time frame	Evaluation of ecosystem status takes place within 20 years after the management measures have been fully implemented.

management scenario, eutrophication management scenario, and the output: ecosystem status. In addition, two auxiliary variables were used to encode the responses of the experts, and the results from the first and second round of evaluation (Fig. 3). This structure enables the encoding of all the experts' opinions on both rounds so that they can be examined expert by expert or pooled together. Each expert's assessments ( $n = 14$ ) were included into the model separately, and each expert was given equal weight. Both the first and the second round evaluations were entered into the model to enable comparison, but the second round results are shown and discussed here.

### 3. Results

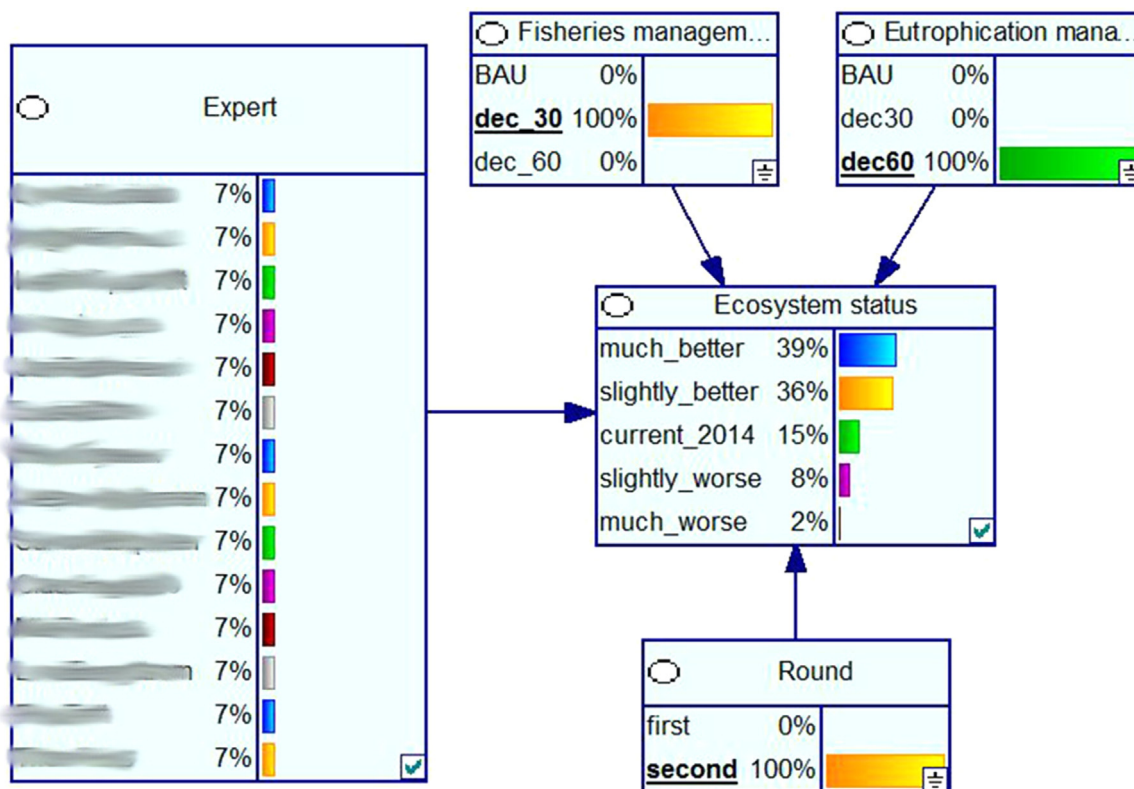
#### 3.1. Spatial model

The reduction of fishing and nutrient inputs in the Baltic Sea resulted in a decrease in cumulative impact scores, and the consequent changes in the impacts were visible (Fig. 4). By applying the threshold of 100, we estimated that the 60% reduction of both

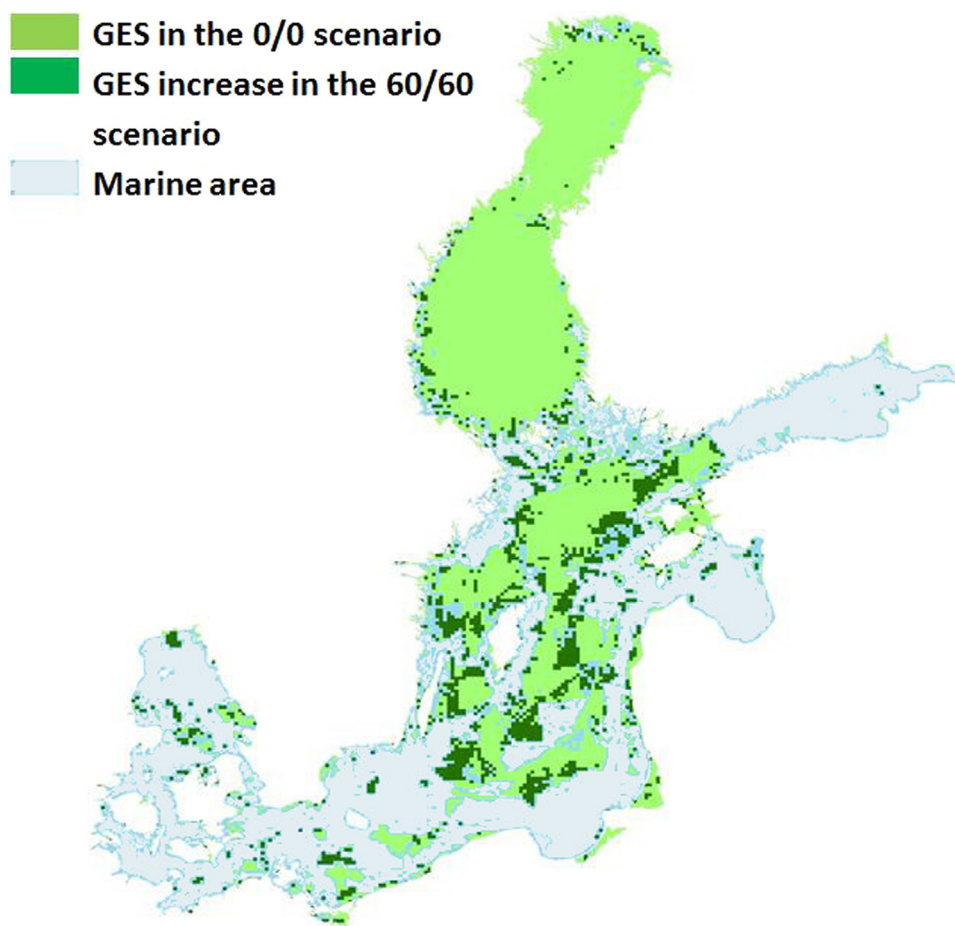
the pressures will increase 19% the area in GES (Table 3, Fig. 4). The predicted impact is greater with nutrient loading than with fisheries management, but joint reductions will have greater effects according to the model. The model did not take into account time lags in recovery and is only a rough estimate of the improvement after the reduction scenarios.

#### 3.2. Food web model

The BaltProWeb model runs suggest that decreasing fishing mortalities result in increases in cod biomass under both nutrient load scenarios (Fig. 5a). Furthermore, higher cod biomasses were projected in the BSAP than REF nutrient load scenarios in every scenario tested, mainly due to the negative effects of hypoxia on cod egg survival. Consequently, the highest cod biomass was projected when the nutrient inputs were reduced (BSAP) and the fishing mortality was at its lowest (-60%). The choice of fishing mortality had a clearly stronger effect on cod biomass than the choice of a nutrient load scenario in the intermediate future (2010–2039) projections. After 2040 (not shown here) the difference in cod biomass between different nutrient reduction scenarios was very small independent of scenario. Also, negative climate effects (i.e., decreasing salinity and oxygen conditions), which are part of the model, caused a decline in cod biomass in all projections after 2030. While cod biomass increases in the scenarios with reduced fishing mortality, the biomass of sprat is projected to decrease regardless the release in fishing pressure (Fig. 5b). This indicates that sprat stock may be controlled by other factors than direct fishing pressure, such as predation by cod and resource availability. The choice of nutrient load scenario had only a small effect on median sprat biomass projections between 2015 and 2039. However, annual variability in the biomass was higher in the BSAP than REF scenarios. Different from cod and sprat, the projections for herring biomass did not show great change



**Fig. 3.** The Bayesian network model structure. In the figure, Fishing – 30% and Nutrients – 60% scenarios are selected from the second survey round results, and all experts have equal weight. In this case, the ecosystem status is assumed to be much better than now with 39% probability, and slightly better with 36% probability.



**Fig. 4.** Predicted area that is in good environmental status when assessed indirectly by the level of cumulative impacts (light green). Predicted increase in the area in good environmental status as a result of 60% reduction in fishery landings and nutrient inputs (dark green). Reductions were expected to be similar over the marine area. Note that in the Baltic Proper, the model does not include the hypoxic seabed as a pressure. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

**Table 3**

Predicted area (km<sup>2</sup>) in good environmental status as a result of the pressure reduction scenarios. The increased area (%) compared with the zero reduction scenario is shown in parentheses.

		Nutrient reductions		
		0%	30	60
Fishery reductions	0	203 175 km <sup>2</sup>	212 125 km <sup>2</sup> (+4%)	224 825 km <sup>2</sup> (+11%)
	30	210 500 km <sup>2</sup> (+4%)	218 450 km <sup>2</sup> (+8%)	231 375 km <sup>2</sup> (+14%)
	60	220 100 km <sup>2</sup> (+8%)	227 350 km <sup>2</sup> (+12%)	242 650 km <sup>2</sup> (+19%)

between different fishing scenarios. Higher herring biomasses were projected in the BSAP than REF nutrient load scenarios (Fig. 5c). These results indicate that the decrease in herring fishing mortality may compensate for the effects of increased cod predation. Also a decreased resource competition with sprat may play a role.

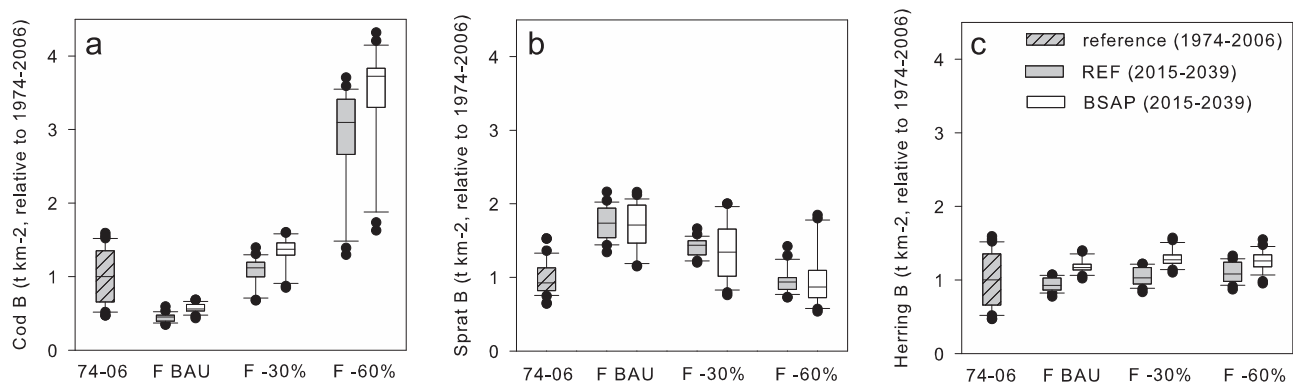
MTL was lower than during the reference period in the F<sub>BAU</sub> and F<sub>-30%</sub> scenarios, but higher in the F<sub>-60%</sub> scenario (Fig. 6). The selected nutrient load scenario seemed to have a smaller effect on the MTL. TST, on the other hand, was solely affected by the nutrient load scenario, indicating that the lower trophic level flows drive TST. In the BSAP scenarios the TST was clearly more variable than in the REF scenarios, making it difficult to make any

conclusions about the nutrient load effects on the TST in the intermediate future. Differences in ascendancy were small between scenarios. Yet, the results show that the relative ascendancy, and potentially food web resilience, is affected by both fishing and nutrient loads (Fig. 6).

### 3.3. Expert assessment model

The results of the expert evaluation showed high uncertainty in the results of the management scenarios (Fig. 7). This uncertainty stems from two sources: first, the uncertainty each expert expressed in their assessments, and second, the disagreement between experts on the strength, and in some cases even the direction, of the effects. The first type, uncertainty about the ecosystem's responses to management, reflects the notion that arose in the group discussions regarding each of the scenarios: there are very many uncertainties in the ecological processes, especially given the extra uncertainty brought in by the climate change. Even though there are records of how the environment has reacted to pressures, the reverse process may not immediately take place as the pressures relax (e.g. Duarte et al., 2009; Nyström et al., 2012). Therefore, there are many plausible but unconfirmed hypotheses about how the ecosystem may react, and the experts' assessments reflected this uncertainty. The second type, the disagreement between experts, mostly stemmed from the fact that the experts' beliefs in the strength of the ecosystem response to management actions varied strongly. This is illustrated by the differences in the





**Fig. 5.** The relative change in (a) cod, (b) sprat and (c) herring biomass between the near future projections (2015–2039) and reference period (1974–2006). The box and whisker plots indicate the median, as well as 25% and 75% quartiles (REF: Reference nutrient load scenario, BSAP: Baltic Sea Action Plan nutrient load scenario, F BAU: Business As Usual fishing scenario, F: fishing mortality, B: biomass).

outcomes each of the experts deemed the most likely for each scenario (Fig. 8). The disagreement was particularly strong in scenarios where nutrient loading was not reduced, but fishing mortalities were, in which case some experts believed in enhancement in the environmental state while others believed degradation to be the most likely outcome (Figs. 7 and 8). Additionally, opinions diverged about what would be the combined effect of –60% fisheries and nutrient reduction management (Fig. 8), because reduced productivity of the ecosystem and reduced fishing pressure might lead to starvation and reduced growth of the fish, with further cumulative effects on the ecosystem structure. In addition, some experts were of the opinion that the fisheries management has a stronger effect on the ecosystem status than the nutrient reductions, while most of the experts believed that nutrient reductions are the more influential management measure.

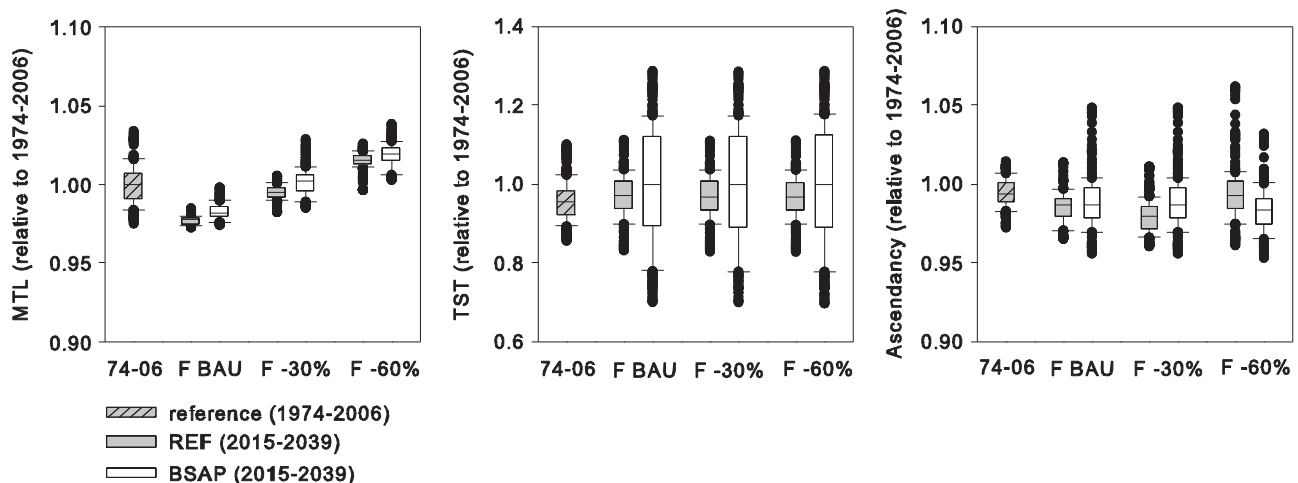
However, despite the large uncertainty, the expert assessment delivered some very clear messages. Firstly, the probabilities of improved environmental status increase with higher reductions of nutrient inputs and fishing. The results also unambiguously indicated that the combined effect of reductions of both pressures were significant: while the probabilities of much better environmental state were 22 and 9% for the –60% scenarios of eutrophication and fisheries management alone, respectively, the

probability of much better state was increased to 46% when both measures were applied (Fig. 7). A central observation was that eutrophication management was deemed to have more potential to influence the state of the sea than fisheries management: the predicted state of the Baltic Sea ecosystem improved more when reducing nutrient loads than when reducing fishing mortality alone, and implementing stringent fisheries mortality reductions without any changes in the nutrient loading policy resulted only in minor shift in the probability distribution towards improvement of the state (top row in Fig. 7).

Secondly, the probability that the state of the sea would improve from the current status was higher than 50% in all the scenarios where anthropogenic nutrient loading was reduced by 60%, or where 30% nutrient loading reduction was combined with reduction in fishing mortality (bottom row and middle and right columns of the middle row in Fig. 7).

#### 4. Discussion

Evaluation of the effectiveness of alternative and concurrent management measures is the first step towards cost-effectively securing the availability and quality of natural resources that in turn provide the goods and services societies depend on. This evaluation also needs to provide reliable estimates of the ecosystem consequences of these alternative management scenarios. In this paper we showed how different management approaches tackle the two



**Fig. 6.** Ecosystem indicators computed by the food web model: (a) Mean trophic level of catch (MTL), (b) total system throughput (TST) (note different scale), and (c) relative ascendancy.

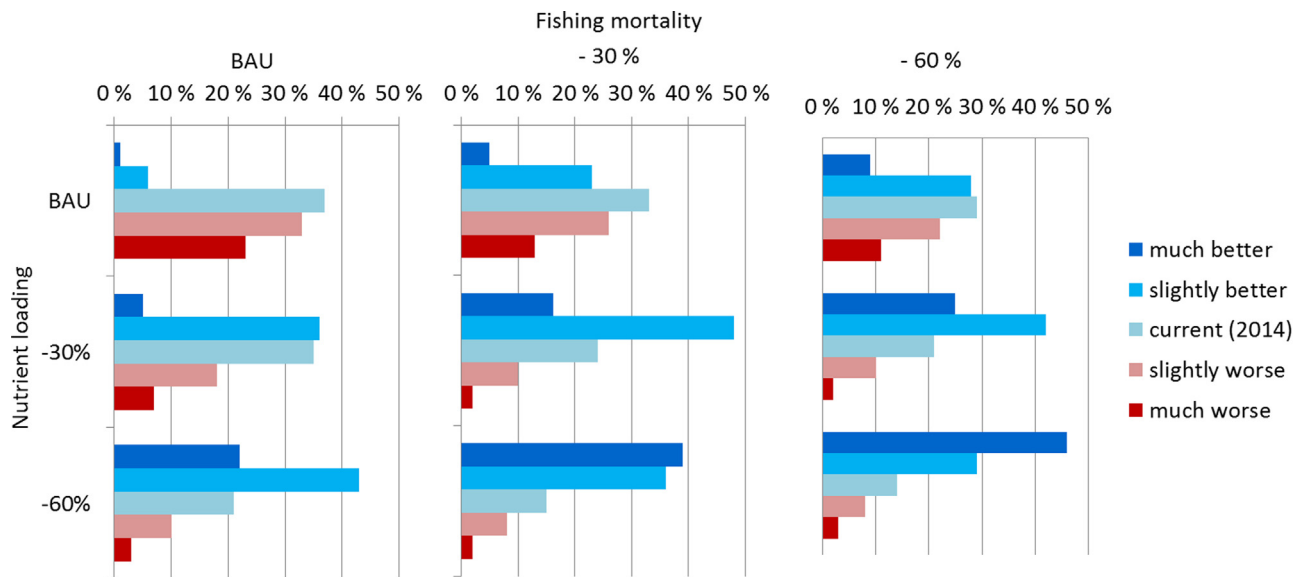


Fig. 7. The resulting probability distributions from the expert evaluation. Each expert's input is weighed equally.

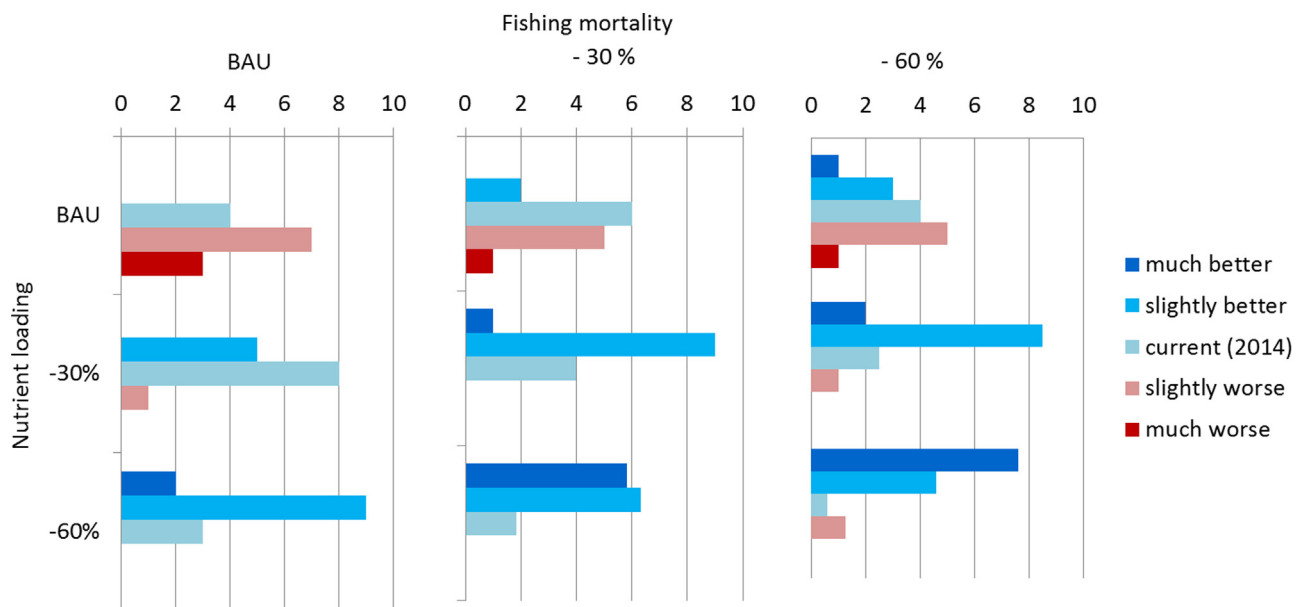


Fig. 8. The most probable outcome of each scenario according to each expert. x-axis shows the number of experts who see each scenario as the most likely one. ( $n=14$ ; ties are split in equal proportions.).

prominent pressures for the Baltic Sea, fishing and nutrient loading. The potential outcomes of these management scenarios (Business as usual, 30% and 60% reduction of pressures) were studied using three different assessment approaches, namely spatial modelling, food web modelling and expert judgement based Bayesian network modelling. All three assessment approaches gave similar outcomes: two models indicated that nutrient reductions produce greater positive effects in the marine ecosystem than fishery reductions, and all models showed that the greatest benefit is reached by joint reductions of these two pressures. Despite the similarities, the three approaches give their estimates in different ecosystem quantities: the spatial model gives predicted spatial increase in GES, the food web model an estimate of change in the species populations, and the expert judgement model the relative improvement of the ecosystem as a whole. Hence, the approaches can be used to complement each other when considering the effects of multiple pressures and what could be the synergistic or antagonistic effects of pressure reductions.

The modelling tools and methods used in this case study operate in different spatial and temporal scales and are also fundamentally different in their approach. Each of these approaches have also specific strengths and limitations which are considered below. Also the implications of these differences for management application and for the further research needs are discussed.

The spatial model assumes additive relations of pressures, lacking estimates of synergistic and antagonistic effects (Halpern and Fujita, 2013), which both have been shown to be prominent in marine ecosystems (Crain et al., 2008). Similarly, recent meta-community studies have shown that spatially mediated interactions between sub-populations of marine species can make it difficult to ascertain clear species-environment relationships (Heino et al., 2015). The simplifying assumptions of the spatial model increase uncertainty related to its outcomes. However, the amount of antagonistic and synergistic effects has been suggested to be almost equally frequent in marine environment (Crain et al., 2008), and therefore the conservative assumption of the additive

effects in our model may be an acceptable intermediate solution. Also the assumption of our model to apply the indicative thresholds for GES can be improved by further evidence for linking cumulative pressures and impacts and the state of marine biodiversity. So far, we were able to base our model on a large scale study in the Baltic Sea (Andersen et al., 2015b). The strength of the cumulative impact model is that it can be used to illustrate geographic distribution of areas where the pressures are many and where the impacts on the overall structure and functioning of the ecosystem and habitats are most apparent. As data on human activities and pressures has become more readily available from many monitoring programmes, increasing our knowledge base, we expect to see improvements in this model in near future.

Some of the main uncertainties in the food web modelling approach are related to future projections that exceed values used in the model calibration, i.e., projecting beyond known space (Dickey-Collas et al., 2014), as well as uncertainties in input data and model sensitivity to choice of prey vulnerability to predation (Niiranen et al., 2012, 2013). Niiranen et al. (2012) identified that these uncertainties can result in varying model results, particularly in long-term projections (see Planque, 2015). However, the type of response (instead of its magnitude) was in most cases rather robust. Also, the food web model is not spatially explicit, and thus will not account for spatial dynamics, such as the recently observed lack in the spatial overlap between cod and its prey sprat (Casini et al., 2014). This lack of spatial information adds uncertainty to the modelling results which assume that all the prey are available to the predators. When projecting ecosystem effects of external drivers certain uncertainty always originates from the scenarios chosen. In case of food web model such uncertainty originates from the global climate scenario chosen, as well as the level of consistency between the climate, nutrient load and perhaps fishing scenarios tested (Meier et al., 2012b). For example, reaching nutrient load reduction targets may be an unrealistic scenario in case climate change will shift more agriculture from South to North. As shown by Niiranen et al. (2013), the chosen scenarios, and their combination, will have an effect on the outcome of the food web model results.

In the setting of the BaltProWeb model no limit for a “good” environmental status is defined, but the model describes biomass change in different food web components. Further, the ecosystem indicators (Fig. 6) describe ecosystem function beyond the mere biomasses of the main fish species, and can be used as an indication of ecosystem health (see also Fulton et al., 2005). However, ecosystem indicator results are dependent of the model configuration (e.g., how many groups are included in the model), and thus setting specific target values is highly challenging. In this study, differences in the ecosystem indicators and fish biomass estimates highlight the need of using both metrics when evaluating ecosystem state. For example, (Fulton et al., 2005) have previously shown that one needs to use a carefully selected suite of ecosystem indicators to capture ecosystem effects of fishing.

Expert knowledge is particularly useful if the variables or scenarios under evaluation are such that there is no or only little data of them (Krueger et al., 2012; Uusitalo et al., 2015), and when integrating elements that are not covered by any single available model (such as coastal and open-sea ecosystems, all trophic levels from microbial loop to top predators, etc.) (Uusitalo et al., 2015). The integrative nature of these models, which makes them useful for management evaluations, however makes them very difficult or even impossible to validate thoroughly, especially if the model parameters are not data-derived (Barton et al., 2008). Proposed ways to validate a Bayesian Network model not based on observational data include sensitivity/information analysis to reveal which variables have most effect on the interest variable (Barton et al., 2008; Fig. 7). In the context of predicting the future, such as

in this study, any validation is by definition restricted to these indirect methods, as the true outcome will only be revealed in the future. The sensitivity analysis of this model gives similar results as the other models: nutrient input reductions are considered more important than fisheries management, but both have a clear impact. However, the results also vary considerably from expert to expert, indicating a lack of consensus opinion or established scientific understanding regarding the effects of the management or our current position along the theoretical response curves.

In addition to the probability distributions, the expert evaluation exercise provided interesting discussion about the knowledge gaps and uncertainties related to the ecosystem and society's reaction to the management measures and consequent changes in the ecosystem. It was particularly noted that there is more uncertainty about how the ecosystem will react if the nutrient loading is not reduced than there is if it will be reduced; further eutrophication and its consequences were seen as negative, but the experts found it difficult to predict what the ecosystem consequences would be. In contrast, the experts found it rather easy to predict the results of continuing with the current fisheries policy, but more difficult to predict how the ecosystem would be shaped if fishing mortality was strongly reduced, either accompanied with decrease in nutrient loading (and therefore overall productivity) or with the business-as-usual eutrophication scenario. Further, the experts noted that stringent management measures would cause major changes in the ecosystem, which would cause changes in human behaviour, which would again have an effect on the ecosystem. As an example of this, improvement in water clarity and changes of coastal habitats would change the structure of coastal fish populations, favouring species that are commercially more valuable (e.g. favour whitefish on expense of cyprinids). This would potentially change the fishing patterns both spatially and in which species they target, which would again have consequences for the fish species and their food webs.

Each of the models have their own strengths and weaknesses as discussed above, and the choice of the best model to support management decisions depends on the exact question that drives the need for management. This exercise, running and comparing the three very different methods, serves as a very good starting point for more detailed management support modelling based on one or several chosen methods. The main message of all the three models was similar, which implies that the basic processes and their directions are well known even if their quantification is difficult. As the three models each take a very different approach, they can be viewed as complementary to each other. While the food web model is specific to temporal trends in trophic interactions of the ecosystem, it is lacking in the spatial aspect of the of the spatial model. The expert assessment can be viewed as a more holistic approach as it integrates the interactions between coastal and open-water ecosystems. Through contradictory assessments and discussion between experts the approach inherently incorporates unknowns and uncertainties within the model domain in a way that no equation-based modelling framework could do. By applying all three types of models, useful information can be attained on both the temporal and the spatial dynamics, as well as on the uncertainties of the related processes. Hence, a more complete picture can be achieved with regard to current state of understanding of cumulative effects on the ecosystem and trade-offs in management decisions.

Baltic Sea countries have recently agreed on joint nutrient loading reduction targets as part of the Baltic Sea Action Plan, and these reductions vary from 0 to 23% total nitrogen and 0 to 60% total phosphorus of the reference inputs of 1997–2003, depending on the sea basin (<http://www.helcom.fi/baltic-sea-action-plan/nutrient-reduction-scheme/targets>). As these targets have been agreed on a high political level, it can be assumed that the

countries view them as feasible. Therefore, the reductions proposed here can be seen as ambitious, but not unrealistic.

Whilst some of the fisheries in the Baltic Sea already exploit below MSY fishing mortalities, some are still too high, relative to the MSY reference points (e.g. cod, sprat, sole). Fisheries science doctrine almost always shows that reductions in fishing mortality to well below MSY reference points result in larger catches, with lower costs and less environmental impact. Thus a reduction in fishing effort of the magnitudes suggested here does not necessarily mean similar magnitudes in the earnings to fishers.

Our results suggest that to effectively reach better environmental state, there is also a need for informed and effective fisheries management. This highlights the necessity of applying holistic and interdisciplinary approaches when defining politically agreed targets on marine ecosystem state.

The evidence of the impacts of fishing on marine ecosystem is strong (Jackson et al., 2001; Daskalov et al., 2007). The fishing impacts on commercial fish stocks have naturally been a subject of research interests and several models have been developed to estimate allowable fishing mortality, either to estimate maximum sustainable yield or to reach a balanced food web (e.g. Michielsens et al., 2008, Möllmann et al., 2014, Voss et al., 2014, Andersen et al., 2015b, ICES, 2015b, 2015c). ICES has given advice on fishing mortality and the state of the commercially exploited stocks for years based on single-species models and is in process to give the advice also by multispecies models (see ICES advice 2014, [www.ices.dk](http://www.ices.dk)). Although the regional models have differences in their predictions (indicating uncertainty in the results), no model of the model ensemble studied by Gårdmark et al. (2013) predicted a recovery of the eastern Baltic cod under high fishing pressure (indicating the robustness of the scientific understanding). Such ensemble model findings of the direction of ecosystem change can be used as indications of ecosystem response to management even when the magnitude or rate of, e.g., ecosystem recovery is debated. The magnitude of the fishery compared to other factors affecting fish mortality was presented by Tomczak et al. (2012), whose model showed that the amount of cod extracted from the ecosystem by fisheries, estimated at over 50% of the annual cod production, was 200–700 times higher in comparison to cod being predated by seals. Also, MacKenzie et al. (2011) found that under future climate conditions fishing and salinity are likely to have a higher effect on the Eastern Baltic cod stock than the potential seal population recovery, as are future management decisions associated with priority for profits in cod fisheries or pelagic fisheries (Voss et al., 2014).

The causality between nutrient inputs and the disturbed environmental status is as strong as the causal link between fishery and fish stocks (and food web). The first quantitative model to estimate the effect of the inputs on nutrient concentrations and other eutrophication related parameters was presented by Wulff et al. (2007) and it has been the basis of the HELCOM Baltic Sea Action Plan (HELCOM, 2007). Reductions of nutrients inputs and fishing aim at recovery of the ecosystem and reaching a specific environmental objective (e.g. the HELCOM vision of a healthy Baltic Sea and the associated quality objectives (HELCOM, 2007)). However, it is likely that the ecosystem will not recover as we expect or the recovery may take longer than anticipated (Refsgaard et al., 2007). Of the three methods in this study, the food web model and the expert panel can consider temporal effects, but the associated uncertainty increases after a few years' prediction and such predictions were not included in this study. Results of the effects of nutrient reduction scenarios on eutrophication indicators in the Baltic Sea have, however, shown that the ecosystem recovery to good status can take as long as 50–100 years due to high sediment reserves of nutrients and complex feedback mechanisms with hypoxia (Vahtera et al., 2007). Although positive

effects of reductions of fishery and nutrient loading have been shown (Aps and Lassen, 2010; Andersen et al., 2015a), the effects of climate change – e.g. increased freshwater outflow, increased stratification and decreased salinity – may have major structural changes to the Baltic Sea ecosystem which jeopardise any long-term model predictions.

## 5. Conclusions and perspectives

In this study we used three approaches to estimate how status of marine environment can change after reductions of two major anthropogenic pressures. Our aim was to test and visualise outcomes of various modelling tools that could be used for advising planning of the management measures for managing multiple actions and pressures on marine environment. As a conclusion we considered all the three approaches useful when implemented together but provide only partial answers when the aim is to evaluate the overall ecosystem state. The methods complemented each other and gave together rather strong support for the need to make big reductions of the pressures but also to consider trade-offs between the two impacting pressures.

The major outcome of this study is the notion that combining different tools for assessing the potential development and change of a marine ecosystem is necessary when planning the management actions and measures. This is particularly relevant when different measures can result in trade-offs between different environmental states. Although the uncertainty in all different approaches is large, models can be used to visualise potential directions of change and thus inform about potential consequences and support planning of management actions. Nonetheless, in order to safeguard the Baltic marine ecosystem and its goods and services, the current understanding gives support for action to (1) reduce nutrient inputs and fishing and (2) give slightly more priority to reduce nutrient inputs, while (3) also remembering that best benefits are achieved through reductions of both pressures.

The models used in this study did not take into account economic costs or social effects, yet evaluating them is also an important part of successful management. Provided that the effects of management measures can be predicted and the associated uncertainty evaluated, economic optimisation of multiple management measures will lead to the desired ecosystem state with minimal associated costs. Taking into account social aspects, such as the commitment and compliance of stakeholders to the management measures, will determine the successfulness of many of the management measures. Therefore, these aspects will need to be tied more and more tightly into the environmental management framework.

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