WP 3.3 Developing tools towards integrated assessment of the marine environment

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List of abbreviations

BAU	“Business-as-usual” scenario
BSII	Baltic Sea Impact Index
BSPI	Baltic Sea Pressure Index
CEA	Cumulative effect assessment
CoD	Cost of Degradation
ESA	Economic and Social Analysis
GE	General equilibrium
GEEM	General equilibrium ecosystem model
GES	Good Environmental Status (MSFD) and good status (HOLAS)
HELCOM	Baltic Marine Environment Protection Commission - Helsinki Commission
HOLAS	HELCOM holistic assessment
IO	Input-output
MSFD	Marine Strategy Framework Directive
PoM	Programme of Measures
SAM	Social accounting matrix
UMW	Use of Marine Waters

1. Introduction

The objective of this part of the SPICE project is to explore ways to operationalize conceptual model of the HELCOM economic and social analyses (HELCOM ESA) developed in the HELCOM TAPAS project The conceptual model includes components for the use of marine waters analysis (UMW) and for the cost of degradation analysis (CoD) and the model was applied to provide regional economic and social analyses for the HELCOM HOLAS II. Figure 1 illustrates the HELCOM ESA components and its relation to the other parts of the holistic assessment. At present, the UMW and CoD analyses are conducted separately from the status assessment and also separately from each other. Integration of economic analysis together with the pressure and status assessment is paramount for the development of economic and social analyses for different marine policies. Marine Strategy Framework Directive (MSFD) calls for cost-effectiveness and cost-benefit analysis of the new measures to achieve good environmental status (GES). Such analyses could also provide advice on achieving economically and ecologically sound marine spatial plans and revision of the Baltic Sea Action plan. Moreover, such integrated model would be useful in the development regional business as usual scenario (see Deliverable 3.3). To these ends, this chapter approaches the development of an integrated model for the HELCOM ESA from two perspectives. At first, we assess the possibilities of applying the Baltic Sea Pressure and Impact indexes in the ESA work (Task 3.3.1). Second, we explore existing literature and projects to provide an overview on theories, methods and applications (Task 3.3.2). Finally, the findings of these two reviews are summarised to provide recommendations for future work (Task 3.3.3).

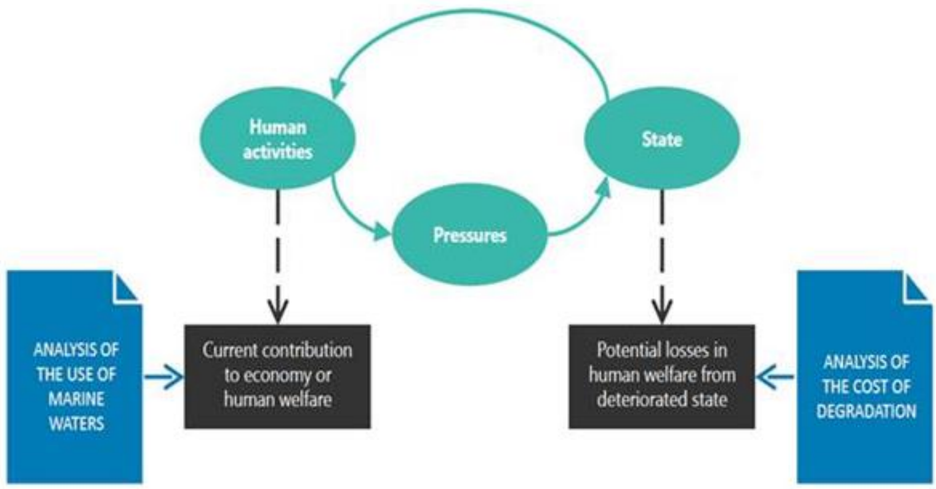

Figure 1. HELCOM HOLAS II economic and social analyses, their relation to the status and pressure assessment and their relation on each other.

2. Assessing the possibilities to use the Baltic Sea Pressure and Impact Index in the regional economic and social analysis

This section gives and overview of the development, current status and paves the way for future work in assessing the links between human activities, the cumulative pressures and impact they cause to the marine environment. Understanding and modelling the causal linkages between the human activities and marine status is paramount for the planning of ecological and economically sound marine policies (Börger *et al.*, 2016; Oinonen *et al.*, 2016b). The recent review of cumulative effect assessment (CEA) methods showed how similar the CEA approaches are worldwide (Korpinen & Andersen, 2016). Majority of the approaches combine spatial data layers of pressures, ecosystem components and sensitivity scores (e.g. Halpern *et al.*, 2008; see Figure 2 for the formula). Also the Baltic Sea Pressure Index (BSPI) and the Baltic Sea Impact Index (BSII), which were made for the HELCOM Initial Holistic Assessment in 2010 (HELCOM, 2010; Korpinen *et al.*, 2012), follow this approach. The BSPI is, however, not assessing the effects but is limited to cumulative pressures, and does not, hence, require ecosystem data.

In evaluation of the applicability of these tools to the regional economic and social analyses (ESA), one should analyse how the tools can handle:

- Ranking most significant pressures and especially human activities affecting the environment;
- Analysing the needs for measures to reduce pressure effects;
- Analysing the effectiveness of measures to reduce the pressure effects; and
- Linking the pressure effects to the state of the environment.

A: Baltic Sea Impact Index

$$I_{\text{sum}}(x,y) = \sum_{i=1}^{n}\sum_{j=1}^{m} D_i(x,y)\, e_j(x,y)\, \mu_{i,j}$$

B: Baltic Sea Pressure Index:

$$S_{weighted}(x,y) = \sum_{i=1}^{n}(D_i(x,y)\frac{1}{m}\sum_{j=1}^{m}\mu_{i,j})$$

Figure 2. The Baltic Sea Impact Index (BSII) and Baltic Sea Pressure Index (BSPI) formulas, where D is the pressure I in a spatial grid (x, y), e is the ecosystem component j and μ is the sensitivity of the ecosystem j to the pressure i.

2.1 Activities and pressures

The BSPI and BSII use the pressure layers as the primary component of the tool. The pressures can, however, be presented by two different methods which have significant differences for the ESA analyses. If a pressure layer is presented as an activity-pressure combination (e.g. Physical disturbance by bottom trawling), the activity information is included in the layer and will be included in the BSPI/BSII outcome. This approach was used in HOLAS I (HELCOM 2010, Korpinen et al. 2012) and in the HARMONY project (Andersen *et al.*, 2013). Alternatively, activity-pressure relationship is identified using a linkage framework and the pressure layer is combined from several activity layers. This approach has been used in the ODEMM project (Knights *et al.*, 2013; Goodsir *et al.*, 2015) and HOLAS II (HELCOM 2017).

The practical reason for preferring the 'combined pressure' approach over the 'pressure by activity' approach, is in the number of sensitivity scores. The latter approach requires much more sensitivity scores, which are set by expert judgment. As these are made for each pressure – ecosystem component combination; each extra score causes a high number of sensitivity scores. For example, an analysis of 10 pressures and 10 ecosystem components requires 100 sensitivity scores, whereas 15 pressures and 10 ecosystem components requires 150 scores.

The BSII analysis is calculated with the EcoImpactMapper software (Stock, 2016) which has the function of showing different rankings of the pressures and ecosystem components. If the pressure layers are 'pressure by activity', then the activity information comes to the rankings and the outcome is directly applicable to the ESA analyses. If the 'combined pressures' are used, one needs to analyse the significant activities separately from the pressure data layers. This approach is slower and thus not recommended.

Deliverable 3.1 on Developing the ecosystem service approach in the ESA framework illustrates how the Baltic sea pressure index can be used to analyse the pressure on ecosystem service as well as where ecosystem services exist and to what extent. In Deliverable 3.1, Figure 1 on p. 12 shows how this approach could be linked with the economic indicators like value added and how such integrated approach could be used e.g. in the use of marine waters analyses. The illustrated approach would be a combination of the ecosystem services approach and the marine water accounts approach.

2.2 Analyses needs for measures

The BSPI and BSII tools can be used to identify needs for measures but also to measure their effectiveness. This however, requires that the significance of the activities is identified in the cumulative pressures (BSPI) or cumulative effects (BSII).

The simple analysis of the need for measures is made in the interpretation of the BSII / BSPI outcome; ranking of the significance of the underlying activities (or pressure by activities) shows the most damaging activities in any assessed area and also shows which ecosystem components are particularly affected. The ecosystem components requiring relief from damaging effects can be chosen on the basis of their status. In other words, the highest ranking activity may not be the most important one for pressure reduction, but the activity causing the highest threat to a species, a habitat or the ecosystem functioning.

The HELCOM HOLAS II assessment presented ranking of most significant pressures affecting the Baltic Sea marine environment. This analysis was made for the entire region as well as for sub-basins. The latter approach is more useful for planning of programmes of measures which should reflect spatial differences in the status assessment. Figure 3 presents the approach graphically.

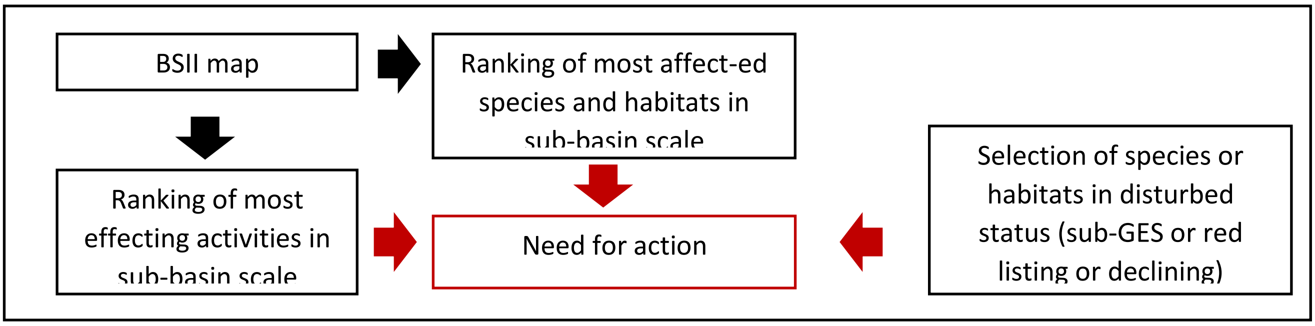

Figure 3. Presentation of the approach to analyze the need to take measures.

2.3 Analysing effectiveness of measures

Effectiveness of the measures is, in its simplest form, a re-analysis of the BSII or BSPI model and comparison of the outcomes. Halpern *et al.* (2015) reran their global human impact model after a few years of the first assessment (Halpern *et al.*, 2008) and made sectorial analyses of the increase or decrease of pressures (and increased or decreased threat to habitats). Such an analysis is very sensitive to changes in input layers; all the other changes than pressure reductions or changes in distribution of ecosystem components decrease the interpretability of the model outcome.

The BSPI and BSII models can be used with scenarios. Uusitalo *et al.* (2016) assumed different pressure reduction scenarios of fishing and nutrients and estimated its effects on the cumulative effects in the Baltic Sea scale. The assumed homogeneous pressure reduction over the entire region, which is likely unrealistic, but a similar scenario-building can be made for smaller assessment areas or with spatially weighted pressure reductions. A good example of the latter would be a pressure layer which is based on a separate spatial model, for example a hydrographical model of riverine nutrient loads spreading in the coastal area. Similarly, pressure reductions can be made with areal protection measures which directly can be included in the BSII / BSPI input layers.

2.4 Linking cumulative effects and the state of the environment

The key limitation is the current cumulative effect assessments is their separation from the state assessments. The state of the marine environment is assessed by selected state indicators, while the cumulative effects are analysed from the pressure point of view (Figure 4). In the HELCOM HOLAS II, the state is assessed by using the HELCOM core indicators and in the integrated assessment also coastal indicators under the EU WFD have been included.

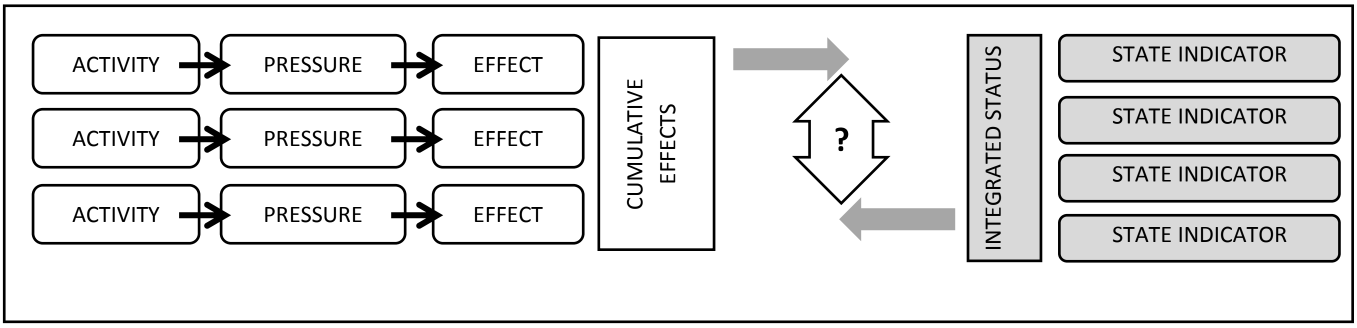

Figure 4. Relation of the CEAs and state assessments. The two assessment approaches give outcomes in different units, but in order to fully use the BSII model in ESA-analyses, the two approaches need to be quantitatively linked.

Finding the causal relationship of state indicators and pressures is a very traditional field of environmental studies. Nonetheless, it is also a difficult issue as many relationships seem to be noisy and unclear. Some causal relationships have been well documented in marine environment, such as the depth extent of macrophytes along the turbidity gradient, state of benthic invertebrates in hypoxic sea areas, or the dependence of fish stocks on fishing pressure (e.g. Krause-Jensen *et al.*, 2008). In contrast, the state of benthic state indicators seems to correlate only weakly with the physical disturbances or eutrophication.

The WP4 of the SPICE project has explored these relationships in more detail, and it is sufficient to only note here that there are data and model-related challenges in linking the BSII effects and the status assessments. This challenge may also be scale-dependent; some links may be more evident in larger spatial scales where more detailed oceanographic or biological variation is levelled off. Andersen *et al.* (2015) reported of significant correlation between the HELCOM integrated assessment of biodiversity status and the BSII and BSPI among nine Baltic Sea sub-basins. The status and the cumulative pressures and effects were averaged over large spatial scales in each sub-basin. Similarly, the SPICE WP4 reported of changes in macrophyte sensitivity index from inner coastal waters outwards; a trend that correlated with the eutrophication status and the intensity of physical pressures.

If the cumulative effects and the state of environment correlate with each other, it is possible to define a level of effects, pressures and activities which should not be exceeded in order to maintain GES in an area. Such a level was tentatively identified with the 2003-2010 data in Andersen et al. (2015) and is also indirectly required by the Commission Decision 2017/848/EU for the assessment of benthic and pelagic broad habitat types. Without the linkage, the CEAs cannot be linked with GES assessments and no definition of 'adverse effects' can be suggested.

Figure 5 illustrates the simple case of one state indicator where the GES gap can be related to the pressure and a reduction in pressure can be calculated. In the MSFD Art 8 assessment, one has tens of GES assessments to carry out and as many GES gaps (cf. the criteria in 2017/848/EU). As the activities, the pressures and the state indicators form an interlinked matrix, where a single activity can cause several pressures and affect even more state indicators, it is not reasonable to plan the necessary measures for each of the GES gaps separately but a more integrated approach would help to avoid over-estimating the need for pressure reductions.

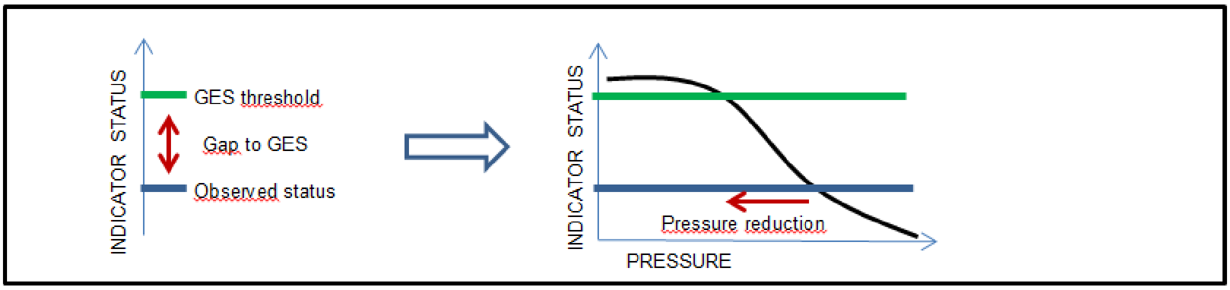

Figure 5. Illustration of the gap to GES concept. Left: an indicator's observed status is related to the GES threshold and the gap can be measured. Right: if the (state) indicator's response to a pressure is known, then the GES gap can be calculated also in terms of pressure reduction.

The current requirement from the Commission Decision defining GES criteria and methodological standards does not encourage the full integration of GES assessments. In practice this means that the GES is assessed for species, species groups, broad habitats, priority substances, some criteria or some descriptors. Altogether tens of assessment results! This does not help defining how far an area, in general, is from achieving the GES or (more importantly to our case) how many pressures and activities are responsible for the GES gap.

To estimate the overall GES gap and the roles of activities behind that, one could follow the steps below:

1. First, collect all the GES assessments in an area. The outcome of this step is a table where all status results and their GES gaps are listed.
2. Secondly, use the linkage framework between activities, pressures and status (e.g. Knights et al. 2013, Robinson et al. 2013, HELCOM 2017). Link the activity and pressure to the GES gap. The outcome of this step is two new columns to the table created above. This linkage framework should also be built in the BSII tool in order to automatize this step and the following ones.
3. Third, organize the GES gaps according to same activities and pressures and estimate which of the GES gaps are partially or entirely caused by the same activity-pressure combination and where the GES gap is independent of other activities. *If an activity is the only responsible for causing one or several GES gaps, then the gap(s) can be bridged (and possibly GES achieved) by managing this single activity. If there are several activities causing the GES gap, then one should analyse the relative roles of each activity-pressure combination preventing GES being achieved.* The outcome of this step is a table where all the GES gaps are organized by the activity-pressure combinations and their roles causing the GES gap are shown from 0% to 100%.
4. The next step is to estimate the necessary pressure/activity reductions. This should be made for each of the activities or pressures separately. Consider the highest GES gap for each activity or pressure (or activity-pressure combination) and decide whether management of this will solve also the other GES gaps related to this activity. If not, make a judgement what would be the extra need.
5. Relate the overall GES gap to the pressure and/or activity (see Figure 5). How great reduction is needed in order to achieve GES?

The BSII can be used to run scenarios of reduced pressures and these scenarios aim to bridge the GES gap or decrease it.

3. Literature review

Both ecological and economic systems are complex, characterized by strong interactions between different elements, feedback loops and time lags (Costanza *et al.*, 1993).A wide variety of external factors, such as species interactions, prices and costs, climate change as well as management settings affect both the ecological and the economic side of the marine system, and thus it is essential to take both the economic and the ecological aspects into account when conducting comprehensive analysis (Prellezo *et al.*, 2012).

Ecosystem conservation indeed is affected by political decisions and by their economic consequences. A change in demand for a particular output impacts on the structure of the ecosystem, and vice versa (Jin *et al.*, 2003). There are many options to model these linkages between economic and ecological systems within an integrated modelling, and next we introduce five of those: bioeconomic modelling, ecological-economic modelling, input-output modelling, and general equilibrium modelling and index-based approaches. Finally, we summarise the pros and cons of the different approaches.

3.1 Bioeconomic modelling

Early bioeconomic applications introduced e.g. a solution to a static fisheries problem (Gordon, 1954; Schaefer, 1957). A dynamic application to fisheries in continuous time was introduced by Clark and Munro (1975). Currently, the number of bioeconomic applications related to fisheries is rapidly increasing, and those have tackled a variety of problems, such as multispecies or ecosystem issues, uncertainty, fisheries agreements and climate change (for reviews on bioeconomic applications to fisheries see e.g. Bjørndal *et al.* (2004) and Kronbak *et al.* (2014)). An example of an empirical bioeconomic study considering Baltic Sea fisheries is by Nieminen *et al.* (2012), which compared the optimal and current multispecies Baltic Sea fisheries management under different environmental conditions taking into account the effects of climate change. The study took into consideration species from two different levels of the food web: the cod as a predator and the herring and sprat as prey species. The biological interactions between the species were taken into account through predation functions. In addition, the biological realism was increased by using an age structure in the population dynamics. The fishery for each species was economically optimised and compared with the current fisheries policy.

Optimal eutrophication management in the Baltic Sea was studied by Iho *et al.* (2016). They used a dynamic bioeconomic model, which considered both the effect of fisheries targeted on cyprinids and nutrient load reduction on the level of eutrophication. Fisheries has a twofold effect on the water quality: The size of the fish stock affects eutrophication through food web effects, and fishing removes nutrients from the ecosystem. They showed that fisheries play a role in cost-efficient eutrophication management in the Baltic Sea.

Albeit bioeconomic modelling is a very useful tool especially when optimising the management of a certain ecosystem component from an economic point of view, it cannot easily cover a wide spectrum of economic sectors or species. This is due to its relatively complex nature, which quickly results in highly demanding optimisation problem. In our example considering fisheries and hydropower, bioeconomic modelling is still suitable as the model involves only two sectors and is thus tractable if the population dynamics is not too detailed. To analyse all relevant human activities and their impacts on the status of the Baltic Sea, bioeconomic modelling would require inclusion of several sectors as well as complex ecological system into the model. Such comprehensive model accounting for the linkages between several economic sectors as well as the ecosystem would require some other modelling approach.

The Blue Growth boundaries in novel Baltic food webs (BONUS BLUEWEBS) project applies bioeconomic modelling to address the capability of the Baltic Sea food webs to provide ecosystem services and by which kind of management measures both the GES and blue growth would be possible to achieve. The project started in 2017, thus the forthcoming project results might be useful in e.g. development of the BAU scenarios or analysing the cost-effectiveness of different management measures at HELCOM or at the Baltic Sea states.

3.2 Ecological-economic modelling

Ecological-economic modelling is similar to bioeconomic modelling, but here we define it to consist of separate complex system models for ecology and economics at their own levels of specificity. Information regarding ecological and economic elements is exchanged between the models and the variables calculated by the ecological model serve as input parameters for the economic model, and vice versa (Jin *et al.*, 2003).

Bockstael *et al.* (1995) is an early attempt to integrate economic and ecological models. They analysed the potential long-term impacts in a specified river basin by integrating a spatial ecosystem model, agricultural components and the decision making process of land-use. The motivation for their interdisciplinary study was to improve understanding how human activities affect ecosystems, what ecosystem services society values and how these values should be measured. A Baltic Sea application of a coupled ecological-economic model is by Ahlvik *et al.* (2014), which incorporated economic data to an integrated catchment and marine model. The dynamic model analyses spatially the least-cost solution of nutrient abatement measures on eutrophication to reach the good environmental state of the sea within a 40-year time span.

Similar to bioeconomic modeling, ecological-economic modelling performs well when the optimal policy of a certain constrained problem, not the balance of the whole economic system, is the desired outcome. However, when the aim is to analyse a system with a large variety of different components, the modelling quickly becomes too complex. Due to this complex nature of modelling, ecological-economic models suit best for problems restricted only to a few sectors. Thus, similarly to bioeconomic modelling, ecological-economic modelling might not be the optimal modelling approach when the aim is to study the Baltic Sea wide activities.

BONUS BALTICAPP Wellbeing from the Baltic Sea – applications combining natural science and economics project applies ecological and economic modelling to assess the fisheries and eutrophication management under climate scenarios in the Baltic Sea. Deliverables of the project include harmonised regional storylines of socio-ecological futures in the Baltic Sea region. Thus in addition to the modelling conducted in the project the storylines could be useful in the forthcoming economic and social analyses at the HELCOM.

Similarly VECTORS of Change in European Marine Ecosystems and their Environmental and Socio-Economic Impacts project studied fisheries and eutrophication management under changing climate. They applied ATLANTIS ecosystem model which is an integrated bio-geo-chemical, food web, multispecies bioeconomic model with a fisheries economics model (Figure 6). The model is able to simulate ecosystem responses to major anthropogenic pressures on fine spatial and temporal resolution.

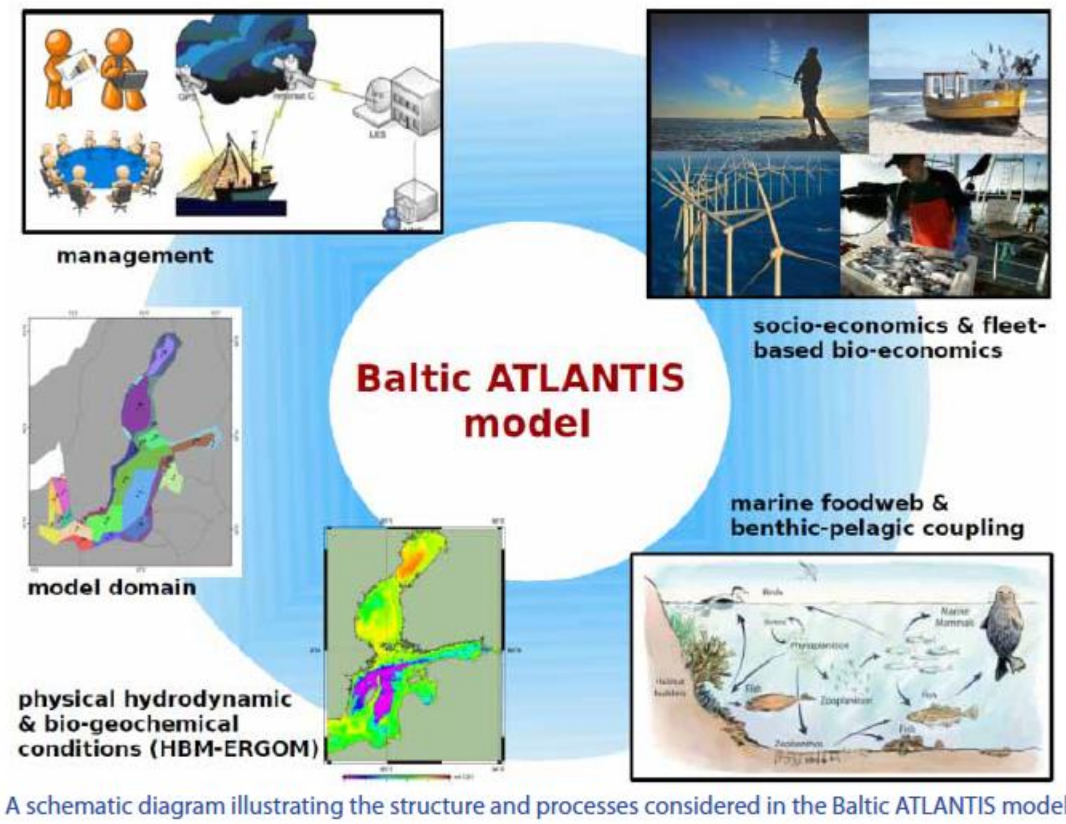

Figure 6. Illustration of the Baltic ATLANTIS model developed in the VECTORS project. (Source: VECTORS Final Report)

3.3 Input-output modelling and its extensions

Input-output (IO) modelling describes quantitatively the linear interdependencies between different economy-wide sectors of a national economy in a matrix format and assesses the policy impacts (Leontief, 1970). The modelling approach has its roots especially on the seminal work by Leontief (1970), as well as on Ayres and Kneese (1969), and it was further applied to ecological processes by Isard *et al.* (1971). IO modelling enables analysing a large number of interacting components, and explains how a shift in demand for economic goods in a particular economic sector affects the output of all other sectors (Jin *et al.*, 2003; Hoagland *et al.*, 2005). IO models explicitly account for all the links across sectors (Jin *et al.*, 2003), and are based on IO table data maintained by nations (Cordier *et al.*, 2017). Thus, as bioeconomic modelling and ecological-economic modelling analyse a partial equilibrium, which isolates the assessed markets from the rest of the economy and thus assumes there are no feedback loops to the rest of the markets, IO modelling considers the economy and as a whole. The same applies to ecosystems, which are comprehensively modelled instead of isolating one or few species from the rest of the ecosystem as is usually done by bioeconomic and ecological-economic modelling. Albeit partial equilibrium modelling can be adequate to understand isolated systems, more comprehensive approaches are needed when the aim is to analyse system-wide effects (Finnoff & Tschirhart, 2011).

This kind of integration of ecosystem with IO model covering several economic sectors is possible as the required national IO table data are relatively well maintained worldwide (Cordier *et al.*, 2017). One useful end product from IO modelling is a map of the links between sectors (Jin *et al.*, 2003). However, IO modelling cannot inform on the optimality of the resource allocation, nor take fully into account the effects of resource depletion or environmental degradation (Hoagland *et al.*, 2005). As IO modelling is only able to consider linear relationships, its applicability to environmental issues, which often represent a non-linear nature, is problematic. Furthermore, IO models are typically able to capture only interactions between economy and ecosystems, but do not reflect the interlinkages inside the ecosystem (Cordier *et al.*, 2017).

Traditionally IO models have been linked to ecological models relatively rarely, but the popularity has been increasing (Cordier *et al.*, 2017). One such example is by Jin *et al.* (2003), who merged an IO model with a linear marine food web model. They showed how changes in primary production in the ecosystem impact on final demand of fishery products. Their application considering one sector and one trophic level can be further extended to cover the full range of economic sectors and trophic levels.

One possible option to overcome the shortage of non-linear nature of IO models based on matrix architecture could be to construct the model the other way around, i.e. to build the IO model into the ecosystem model as being one component of the ecosystem, as is done by Cordier *et al.* (2017). They applied such method to fish habitat and harbour development, and modelled the interacting dynamics of the systems over time taking into account the feedback loops and time delays, thus reducing the static nature of traditional IO models. Their approach allowed estimating the indirect economic consequences on one sector caused by ecosystem changes impacting on another sector. Furthermore, their method is able to analyse the winning and losing activities through identified trade-offs between sectors (Cordier *et al.*, 2017).

Another extension for IO modelling is the social accounting matrix (SAM), which can be used for policy analysis. SAM is an established technique that combines IO table data with national income accounts. As IO captures only interactions among sectors, SAM incorporates the interlinkages among production activities and factors, income and consumption, as well as capital accumulation in an accounting framework (Xie, 2000). Thus, SAM is basically an extension of IO models that considers all monetary flows in an economy (Finnoff & Tschirhart, 2011). Similar to IO modelling, also SAM has traditionally neglected the environmental aspects. However, Xie (2000) introduced a concept applying interactions between economy and environment within a SAM approach.

As IO modelling considers the whole economy, it fits well for linking the wide variety of sectors using marine waters with the actual status of the environment. The challenge applying IO modelling is that so far there are not too many examples linking it to ecology. Furthermore, ecological part would likely need to be simplified, as IO models cannot consider non-linearities. Despite these slight challenges and compromises, IO modelling might offer a valuable tool for integrating marine ecosystem with the economic sectors utilising it. In our example considering fisheries and hydropower energy sectors we can think the example as follows: Fisheries, affected by fish production and its final demand, creates jobs and income for households, and simultaneously supports other sectors that provide inputs for fisheries, such as boat-building, repair and energy. When the links of energy sector as well as of all other industries are identified, as now we can consider the economy as a whole not restricted only on two sectors, income and employment effects of the region generated by the sectors can be revealed.

To our knowledge, input-output modelling has not been applied in the Baltic Sea region to address marine policies. Regional Adaptation Strategies for the German Baltic Sea Coast (RADOST) project used ecological model together with input-output model to assess the effects of climate change and adaptation strategies on regional income and employment. However, their analyses was limited to the agricultural sectors and the results were published mainly in Germany.

3.4 General equilibrium modelling

General equilibrium (GE) modelling represents mathematically the economy as a complete system of interlinked elements, such as households and industries, demand and supply as well as imports and exports. It takes into consideration that an effect of economic shock impacting on one component can have effects on any other component and thus the whole system (Dixon, 1992). GE has many features in common with IO modelling, such as questions addressed and data requirements. However, GE can be seen as an improvement over IO modelling due to a broader range of interactions, non-linearities involved and substitution possibilities (Rose, 1995), as well as to the sequence of static optimization and resulting equilibria (Finnoff & Tschirhart, 2008). Furthermore, GE relaxes some unrealistic assumptions used in IO modelling, such as fixed prices (Finnoff & Tschirhart, 2011). In general, GE modelling assesses policy impacts on standard economic indicators (Banerjee *et al.*, 2016). Similar to IO modelling, also GE modelling analyses the equilibrium by considering the economy and as a whole (Finnoff & Tschirhart, 2011).

As in the case of IO modelling, only a few studies have linked GE to ecological models so far, albeit the interest has been increasing (Banerjee *et al.*, 2016). However, such integration is possible as economies and ecosystems have similarities both being non-linear complex systems with extensive interdependency between elements (Arrow, 2000). One example combining GE with marine ecology is by Finnoff and Tschirhart (2008), who consider commercial fishing and tourism and introduce an application of an economic GE modelling linked to a general equilibrium ecosystem model (GEEM). GEEM is a method that combines GE calculations with population dynamics (Finnoff & Tschirhart, 2011). Similar to IO modelling, output variables from the GEEM model are used as input parameters into GE, and vice versa. Linking GE with GEEM can be useful for identifying the most important interconnections between economies and ecosystems (Finnoff & Tschirhart, 2008).

Banerjee *et al.* (2016) introduced a conceptual framework for integration of economic and ecological data by combining GE and System of Environmental-Economic Accounting (SEEA). Their novel quantitative approach enables a comprehensive analysis of policy impacts on both the economy and the environment. Another ecological application of GE is by Finnoff and Tschirhart (2011), which combined GE economic and ecological models through fisheries and agricultural runoff, and increased the accuracy of the model by applying nonlinear interlinkages inside the ecosystem. Further marine-related GE applications are e.g. Jin (2012), who studied the aquaculture and fisheries sectors, Narayan and Prasad (2006) focusing on fisheries export strategies and Seung and Waters (2010) assessing the impacts on the economy if catches, fuel prices or seafood demand change.

GE modelling is a tempting approach as it has the positive characteristics mentioned earlier considering IO modelling, but it overcomes some of its shortcomings, such as substituting linearity with non-linearity. However, as GE aims at presenting the system more realistically, it requires a vast amount of detailed data as well as extensive modelling capacity and other resources.

To our knowledge the VECTORS of Change in European Marine Ecosystems and their Environmental and Socio-Economic Impacts project is the only attempt to apply GE modelling in the Baltic Sea to assess macroeconomic consequences of marine ecosystem changes. Unfortunately, the results are reported in an unpublicized deliverable and the project final report (p. 40) only gives a one page summary.

3.5 Index-based approach

Ecological indicators and indices were initially developed to consider the impacts of dominant pressures. However, the recent trend has expanded to inclusion of socio-economic and governance issues of multiple human activities (Coll *et al.*, 2016). Index-based approach is a simple and pragmatic way to combine the existing knowledge on economy and ecology when data limitations exist.

Hoagland and Jin (2008) increased understanding on ecological conditions of marine environment and its economic value by comparing the developed intensity index of human activities in the marine areas with a socio-economic development index. They identified regions that could have the potential to achieve sustainable development of the marine environment on their own and those that may need international financial assistance to achieve the target. The marine industry activities were measured by a set of indices, and it was assumed that higher level of activities caused greater pressures on the marine environment. Furthermore, with a given level of activity, the pressure on the environment was dependent on each nation's stage of the economic development. To assess the level of activity, data considering following aspects were used: fish landings, aquaculture production, shipbuilding orders, cargo traffic, merchant fleet size, oil production, oil rig counts and tourist arrivals. The index could be further developed by including e.g. data on population density, coastal development and levels of pollution, albeit collection of such information would need further effort.

Cheung and Sumaila (2015) developed a spatial bioeconomic vulnerability index of fish stocks to assess overfishing globally. The derived index is based on economic discount rates and biological intrinsic population growth rates of all major exploited fish species in the world. The index is helpful in identifying regions that are sensitive to overfishing in the absence of effective fisheries management. They conclude that with low bioeconomic vulnerability and high intrinsic growth rate it is possible to achieve a 'win-win' solution between economic benefits and conservation of marine environment. Another ecological marine-related index considering socio-economic aspects is by Halpern *et al.* (2012), who created an ocean health index assessing the healthy human-ocean system of all coastal countries in the globe. The index comprises ten public goals, including coastal livelihoods and economies, and it is based on the current status as well as on the likely future status of the marine environment. The future status is based on e.g. the pressures each goal faces, which are further divided into ecological and social pressures. Ecological pressures include pollution, habitat destruction, species introductions, fishing and climate changes, and social pressures consist of e.g. poverty, political instability and corruption.

Index-based approach could be applicable in the context of the Baltic Sea, as the existing Baltic Sea Pressure and Impact Indices (BSPI/BSII), similar to the activity index developed by Hoagland and Jin (2008), could be utilised. BSPI/BSII could be compared to socio-economic indices, such as value added and employment of each sector, and the relationship between the indices could be identified. Such approach would be able to answer the following question: If human activity X increases/decreases by Y%, how much it would affect the BSPI/BSII and furthermore to the socio-economic indicators of all activities? This approach would link the marine activities to the status of the sea, and to other activities as well, and would thus require an estimation of activities' dependencies and sensitivities on the status of the sea (See also Deliverable 3.1, p. 10) BSPI and BSII could be further developed e.g. by taking into account the sustainability of each activity. The index-based approach does not inform anything about the optimal policy, but can instead be used to assess the implications of different development scenarios of the activities in the Baltic Sea. If the index-based approach was conducted spatially, it would give valuable input for marine spatial planning.

An example of index-based approach is STAGES was a Coordination and Support Action, which received funding from the European Union under the Seventh Framework Programme of Cooperation. Under the STAGES project, a set of management scenarios with the BSII tool was run in order to study various alternatives to increase our knowledge about the management effectiveness in marine environment. The study assumed fisheries and eutrophication reduction scenarios where all nine combinations of 0%, 30% and 60% reductions were applied. The BSII tool was used to calculate cumulative impacts on the Baltic Sea ecosystem with the reduction scenarios. The results showed great improvement of the status especially in the areas where these two pressures were dominant (such as the SW Baltic Sea). The study also found a significant dependency between the cumulative impacts and the state of biodiversity (as defined in Andersen et al. 2014). Using this dependency, the authors defined a threshold value which was assumed to represent the maximal cumulative impacts (similarly a threshold for maximal cumulative pressures from the BSPI was identified). By this threshold it was possible to compare the scenario outcomes and see how much the area in GES would increase after pressure reductions. Rather surprisingly, the GES increase occurred in areas where the pressures were already rather low and the heavily pressurized areas (e.g. SW Baltic Sea) did not achieve GES even under the highest reduction scenarios. The outcome was published by Andersen et al. (2015) and Uusitalo *et al.* (2015).

3.6 Summary

Figure 7 summarises the reviewed approaches that integrate ecological and economic aspects related to marine management. The reviewed approaches are suitable for different types of marine policy analyses and they vary in terms of details considered in modelling the economic and ecological systems, how time can be considered and how much resources are needed to set up a model and run it. Fisheries and eutrophication management in the Baltic Sea has been studied in several projects using bioeconomic and ecological economic modelling. However, the other approaches have not gained that much attention. The use of models or index based approaches in the support of MSFD economic analyses seems to be lacking do to the lack of suitable models and data (Börger *et al.*, 2016; Oinonen *et al.*, 2016b). Thus, the economic analyses related to the MSFD programme of measures are based on expert assessments (Kontogianni *et al.*, 2015; Börger *et al.*, 2016; Oinonen *et al.*, 2016a). To our knowledge the use of existing biological models (Tedesco *et al.*, 2016) together with economic data and models have not been assessed.

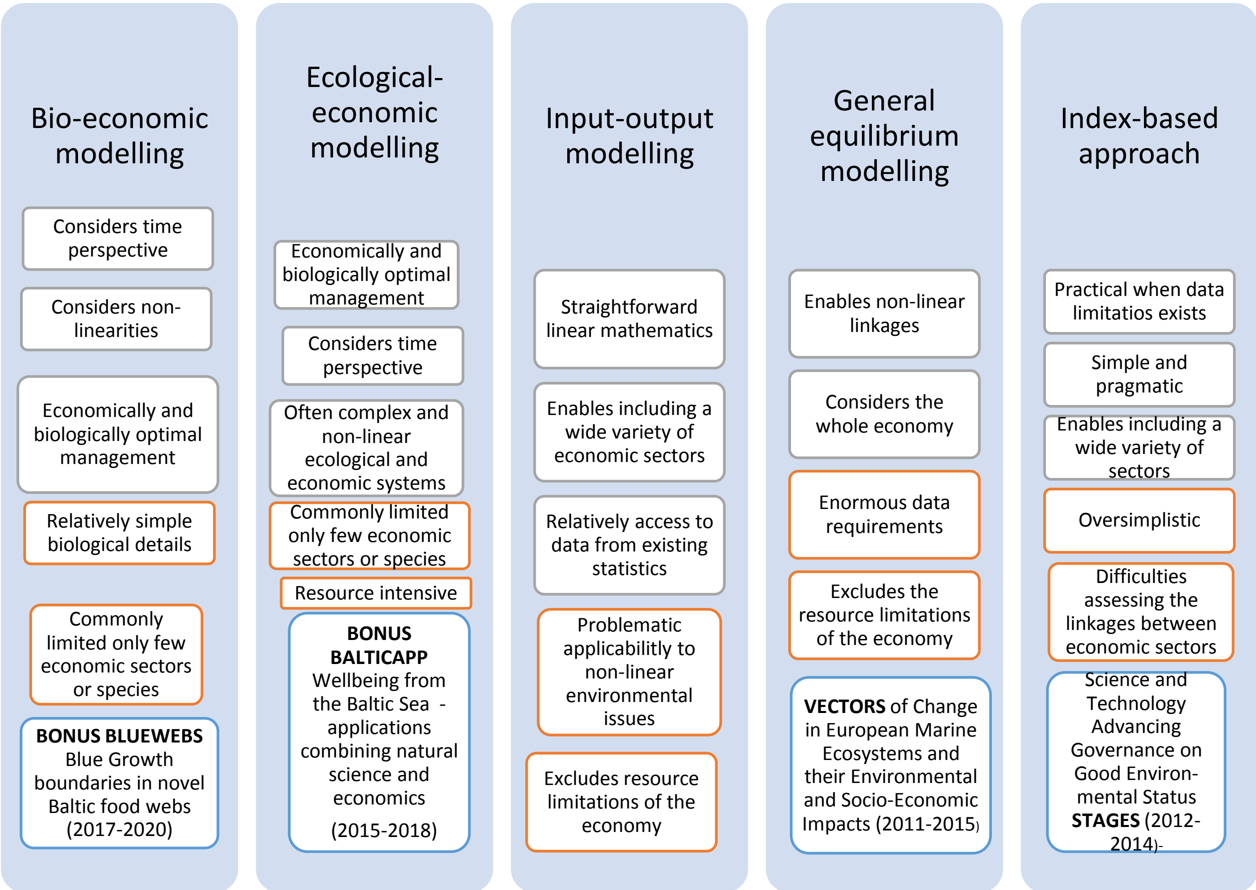

Figure 7. Summary of the reviewed integrated approaches and their pros (green box) and cons (red box) and examples of projects (blue box).

4. Recommendations

The SPICE recommendations to improve the economic analyses and especially support the planning of programme of measures are outlined below.

1. Modify the BSPI to support the economic analyses and programme of measures (PoM): The tools would better support the PoM if they had built-in functions to relate activities to pressures in correct quantities. At the moment the tools include only pressure data and the underlying activity data (that causes the pressures) are managed outside the tools. With the built-in activity-pressure relations, the tools could calculate the amount of specific activity causing a specific impact in an area. This amount would, obviously, depend on the quantitative relations included in the tool.

2. Research activities needed to link state and pressures: The greatest gap limiting the use of BSPI/BSII tools is the difficulty in relating the state assessments with the pressure assessment. Positive examples were reported above and more examples were discussed in SPICE WP4. Figure 4 presents the basic problem. This challenge is even more important as the new GES criteria from the 2017/848/EU require defining 'adverse effects' from a wide set of pressures; a task that cannot be made without relating the pressures to states. Obviously, more research focus should be given to the temporal and spatial scales of pressures and states which might help in finding robust links between them.

3. How to analyse effectiveness of measures? The aim of measures is to improve the state of the sea. Such an improvement can be assumed to occur if a pressure is decreased. However, to observe change in the state of the sea, one should have time series data and sound understanding that the state indicator responds reliably to the change in the pressure.

4. How to plan effective measures? Planning of effective measures is a predictive approach which must be built on models. The cumulative effect assessment models were discussed above and it was noted that in order to plan an effective measure one must know the gap to GES. This gap can be specific to a species, species group, substance or a descriptor, or it can be an overall gap as described above. In this latter case, one assumes that the greatest GES gap in a state indicator would be sufficient to bridge the GES gaps of other indicators as well. This may not always be the case and, hence, some expert judgement is needed to solve this. The amount of pressure or activity reduction is predicted from a state-pressure curve (see Figure 5).

5. How to accommodate uncertainty into the models? The cumulative effect assessment models have previously not considered uncertainties. First steps towards considering uncertainties in the assessment could be taken following Stelzenmüller *et al.* (2018).

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