Elsevier Editorial System(tm) for Ocean & Coastal Management
Manuscript Draft

Manuscript Number:

Title: Integrating ecological, economic and social aspects to generate useful management information under the EU Directives' 'Ecosystem Approach'

Article Type: SI: Wadden Sea

Keywords: strong and weak sustainability, integral system, habitat, ecosystem functioning, ecosystem processes, ecosystem services, network analysis, DPSIR framework, valuation

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To: The Editor-in-Chief of Ocean and Coastal Management
Date: 22nd May 2011
Subject : Cover Letter

Dear Editor,

On behalf of the two co-authors Rute Pinto (University of Coimbra) and Kerry Turner (University of East Anglia) I hereby submit our contribution to the OCMA Wadden Sea Special.

The paper title is:

Integrating ecological, economic and social aspects to generate useful management information under the EU Directives' 'Ecosystem Approach'

This contribution is basically a 'vision' paper describing a strategy of how to bring the ecologists and the enviro-economists together in the arena of the decision making process where we have to judge both the ecosystem and the economic system based on which decisions are taken. Any decision about the Wadden Sea should contribute to 'sustainability' in terms of system development as well as welfare or well being.
Based on our work we strongly plea for following the line of the 'strong sustainability' where substitution rules are not only focussing on substitution of natural capital by monetary capital but where other rules (e.g. 'precautionary principle') play a vital role in conserving the area and its values so that also future generations can profit from it.

Based on our findings we are the opinion that the integration of ecology and economy should be made concrete by applying 'network analysis techniques' at the habitat level because that is a suitable level to designate both human functions as well as ecosystem functioning and properties. As a consequence, the only fruitful way forward is a holistic approach (where all the habitats together represent the ecological part of the integral system consisting of the combination of the total ecosystem and the total economic system) which is in line with the recommendation for an ecosystem approach by the EU Commission via the Water Framework Directive and the Marine Strategy Framework Directive.

We seek publication in the OCMA Wadden Sea Special as we find that researchers, policy makers, managers and politicians and in discussion with the Wadden Academy need to define how to schedule and realize the so-called "Wadden Sea Knowledge Agenda".

Regards,

Prof. Dr. Victor N. de Jonge (DSc University of Hull)

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Integrating ecological, economic and social aspects to generate useful management information under the EU Directives' 'Ecosystem Approach'

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***Highlight**

Research highlights:

Integrating ecological, economic and social aspects to generate useful management information under the EU Directives' 'Ecosystem Approach'

- Ecological and economic information should be integrated at the habitat level
- System valuation is only possible when a holistic approach is applied
- The DPSIR (driver, pressure, state, impact, response) framework is a helpful technical tool here
- Future application and further development of (Ecological) Network Analysis techniques for management purposes is strongly recommended

ABSTRACT

If we as scientists can not decide upon what research, monitoring and technical tools should be used as a basis for policy making and management, then the politicians and other decision makers will continue to follow the line of ‘weak’ sustainability (applying monetary substitution rules to natural capital) instead of ‘strong’ sustainability (applying alternative rules like the precautionary principle). Suitable integral indicators or indices covering ecological as well as socio-economic aspects are thus required. There is, however, a clear friction between what can be delivered in terms of useful ‘(integral) indicators’ and what decision makers require us to deliver in terms of ‘simple, cheap, easy to understand’ while the real situation is extremely complex. This social, economic and ecological complexity has been an important impediment to the required technical co-operation between the decision makers and the natural and social scientists since the publication of the Brundtland report. Given the panarchic character of natural systems realistic base environmental indicators should be anchored to a thorough examination of the functioning and the structure of ecosystems and related integrated indicators instead of the use of dynamical models deficient in reducing the uncertainty as to future system behaviour, or selecting for ‘cute and cuddling’ icons of any ecosystem without knowing what they ecologically represent. The connection of the social and the ecological aspects in an integrated approach is thus pivotal to make sustainability as starting point a ‘reality’. To arrive at the required integration we propose that decision makers should stop asking for ‘simple’ environmental indicators and accept the complex reality that is our environment. To achieve this we propose that they should buttress to make the Odum food web ideas functional by the application of ecological network analysis (ENA) at a scale where socio-economic and ecological information can be integrated, which is the ‘habitat’ level. At the habitat level ecological functioning (natural compartment), human activities (economic compartment) and ecosystem functions to humans (socio-ecological compartment) can be designated and measured. This process can further be facilitated by the use of the Driver-Pressure-State-Impact-Response (DPSIR) approach. To facilitate the weighing and decision support systems’ process we propose to apply multi-criteria techniques to integrate all the information. In practice the adaptive management process might be a suitable option. As a consequence it is crucial to define what to investigate, what to monitor and how this subsequently can be related to the relevant sectors of the economic part of the integral system to realize sustainability in line with the Brundtland Commission’s view.

Key words: strong and weak sustainability, integral system, habitat, ecosystem functioning, ecosystem processes, ecosystem services, network analysis, DPSIR framework, valuation

1. INTRODUCTION

1.1 Sustainability, the ecosystem approach, ecosystem services and natural capital conservation

In 1987 the Brundtland report was published with a definition on sustainable development (SD) "*sustainable development meets the needs of the present without comprising the ability of future generations to meet their own needs*". It is one of the most widely used definitions but it stresses high level socio economic goals rather than a working blueprint for sustainable science, policy and practice within an integrated system. Ten years later she (Brundtland, 1997) stated: *"in ocean management, as in most other areas of human endeavour, close co-operation between scientists and politicians is the only way to move forward. Science must underpin our policies. If we compromise on scientific facts and evidence, repairing nature will be enormously costly, if possible at all."* This statement is the basic starting point of the present contribution in which we will try to set out practical directions on 'how' to approach sustainability.

In the field of governance the SD definition was refined in economic terms into 'weak' and 'strong' variants (Turner, 1993), but SD was also considered by some analysts to be still fuzzy and ambiguous as an integrative real world strategy (e.g. Custance and Hillier, 1998). This served to inhibit its practical and immediate implementation. To overcome this drawback a set of guidelines was developed (Bellagio principles) to reinforce the holistic perspective within SD (Hardi and Zdan, 1997). The guidelines focused on whole system accounting which encompassed the well-being of social, ecological and economic components.

From the governance perspective, environmental legislation has become more extensive and the expansion of the European Union, for example, has led to a situation where all member states implement uniform EU Directives (subject to subsidiarity clauses) with the common goal of environmental protection. Potentially this has led to an increase in the harmonisation of the environmental legislation among the EU member states. An important and valid question is, however, whether the practical implementation at the different national levels in practice, also satisfactorily fulfils the starting points and goals of the relevant EU Directives. Attempts to achieve a balanced and sustainable utilisation of natural resources (either at a global, regional or local scale) cannot be adequately considered as a sectoral, social, or economic problem in isolation. Thus, any action (e.g. future technological switching and other adaptation measures) requires a better understanding of the complex interactions between all parts of the 'integral system' (Fig. 1). Sustainable development requires at its core a fuller appreciation of the long-term impact of the increasing scale and rate of human activity on the environment (Hardi and Zdan, 1997).

[insert Fig. 1 here]

Since the Brundtland report the need for a more comprehensive monitoring of societal and environmental development impacts is widely acknowledged as being an important source of information to the authorities involved in economic growth and wealth creation promotion, and those involved in the management, conservation, and protection of the environment. Apart from this formal role, monitoring is, however, also important in feeding data to the scientific community and in informing society in general and influencing social norms. Despite the fact that the Brundtland report was published 25 years ago, monitoring of the ecological and socio-economic impacts of environmental change has been largely confined to defined sectors, because of lack of a generally applicable interdisciplinary conceptual and analytical framework. Consequently, current monitoring programs are still carried out in isolation from each other while we know and acknowledge that we are dealing with 'social-economic-ecological' or 'integral' systems and connected linkages (Berkes et al., 2003) (Fig. 1). Societal developments such as changes in land use (urbanization, industrial developments and agricultural practices) climate change and the environmental and social feedbacks to these changes are connected in a complex way. This complexity has been an important impediment to the required technical co-operation between the political process and natural and social scientists that Brundtland called for. More recently there has been a growing awareness in the decision making process of the findings from behavioural sciences which highlight among other things the importance of networks and often complex linkages in human behaviour and change over time. Legal instruments, for example, are now being buttressed via so-called 'nudge' policy measures with more subtle motivations (Layard, 2010; Ormerod, 2010). But complexity is equally a characteristic of natural systems of which humans are a part (Berkes et al., 2003). The need for more comprehensive environmental accounting frameworks has never been greater. Nevertheless, our decision makers continue to call for environmental indicators which are 'easy to understand' and 'cheap and simple to measure' despite the fact that ecologists have yet to agree a common integrated concept of the most relevant basic processes responsible for the natural panarchy in expressions of one and the same geographical system as visualised in Fig. 2 (see also de Jonge, 2007; de Jonge et al., 2006). The existence of multiple biological system expressions is due to the many natural variations in the physical and physico-chemical boundary conditions represented by e.g. varying conditions in wind, irradiance, temperature, salinity and nutrients. Strong winters may for instance lead to mass mortality of intertidal (benthic) fauna (Beukema, 1985) which changes the 'top-down control' by a strongly reduced grazing pressure on micro-algae in the water column and on the sediment. However, dull weather conditions during summer may negatively influence the 'bottom-up control' by decreased primary production. According to the resource competition theory (RCT) varying resource conditions (nutrient concentrations and light conditions) affect the abundance among species (e.g. Grover, 1997; Huisman and Weissing, 1999, 2001; Tilman, 1982); while according to the intermediate disturbance hypothesis (IDH) (Connell, 1978; Horn, 1975) the development of the species structure may be directed to a structure deviating from the common one. Moreover, invading species may occupy either a new niche or may simply replace (part of) autochthonous species at different trophic levels. On top of this, human activities affect the system for instance by fishing (damaging biological population structures and habitat), dredging (increased turbidity and habitat destruction), and discharges of waste water (loading of the system with pollutants and nutrients). There is thus not only a strong inter-annual variation in the abundance of individual species but there may also be a significant inter-annual variation in the structure of the system expressed by species composition and abundance (Fig. 2).

It is not possible to decide objectively which expression in nature is wrong, acceptable or good because objective criteria are not available for our dynamic coastal systems (see also Elliott and Quintino, 2007). The above suggests that from an ecological point of view monitoring should preferably be focussed on integrated indicators which tell us something about the functioning of the entire food web, i.e. the combination of structure and functioning, instead of solely indicators reflecting parts of the system's condition (see Naeem et al., 2009). Such strategic decisions are necessary for an effective and efficient conservation of our living environment. When we, as scientists, are not competent or not able to decide in a coherent and reasonably unified way what to champion to aid decision making in terms of research and monitoring and effective policy measures, then the politicians will often be 'persuaded' to take action (or no action) on the basis of short term or overly 'local' considerations. The long term sustainability of both the natural systems and the wealth creation potential of the ecosystem services they ensure may not get the required recognition with negative consequences for natural systems and human wellbeing (Boyd and Banzhaf, 2007; Fisher et al., 2009). Kremen et al. (1994) state that conserving nature is only possible when it is combined with attention for the wellbeing of the local population something we fully agree with because it underpins the importance of connecting the socio-economic, cultural, political and the ecological aspects in an integrated approach so that sustainability becomes reality.

[insert Fig. 2 here]

Interestingly, during recent years, the European Commission has published a number of Directives aiming at an integrative approach when assessing the quality of the natural environment. Important directives here are the Water Framework Directive or WFD (EC, 2000) and the Marine Strategy (Framework) Directive (EC, 2005a, b; EC, 2008). In one or another way these directives follow the integrative principles of 'systems ecology' and those of the 'ecosystem approach' or EA (de Jonge, 2007). This consequently means focusing on the ecosystem part of the integral system (Fig. 1; see also Likens, 1992). Environmental conditions should be assessed on the basis of the structure and functioning of the biological part of the ecosystem in response to the sum of the natural variation (caused by natural stress factors) and the human induced stresses. This relatively narrow focus (only on the quality of the biological expression of the ecosystem) has emerged because of the recent rapid deterioration of some of our environments and because decision makers have given broad attention mostly to the socio-economic part of the 'integral system' or 'social-ecological system'. But even this attention needs to be further enhanced if a strong sustainability position is accepted.

Strong sustainability requires that among other things, ecosystems are seen as suppliers of a range of intermediate and final services (ecosystem services approach) through which humans benefit in terms of welfare. Sustainable utilisation of this vital resource base is therefore the key notion. It can be argued that the assigning of monetary values to the benefits provided by 'healthy' ecosystems can supplement scientific and ethical arguments in favour of environmental protection and biodiversity conservation (Turner et al., 2010).

The WFD, within the present context discussed by de Jonge et al. (2006), distinguishes two simple complementary ways of reaching its goals:

1. the optimization of the physical habitat-providing conditions; and
2. the (further) improvement of water quality.

The result of this should then be assessed by the quality of the structure and the functioning of the biological part of the system. Assessing the biological quality of estuarine and coastal waters is any case a difficult task because of the variability of these systems (e.g. Elliott and Quintino, 2007; de Jonge et al., 2006). The available benthic macrofauna related biological indicators turn out to be non-comparable with each other, which may indicate that they are unsuitable to assess the quality of the biological structure under consideration because they all cover a different aspect of the ecosystem part under consideration (Patrício et al., 2009). Finally and within the given political context any assessment of the biological quality is also not a simple task because (see also above) politicians continue to call for 'simple, easy and cheap to measure' indicators. There is thus a clear friction between what can be delivered at the moment and what is called for by the decision makers. The results of any assessment should also be meaningfully connected to any (natural or human induced) stressor or set of stressors to provide effective indicators and an effective human response to the new situation.

To sum up so far, and using the EU Marine Strategy Framework Directive as an example, policy needs to be 'informed' by the 'ecosystem approach' and the 'good environmental status' needs to be interpreted in terms of ecosystem structure and functioning plus services provision. Implementation of the approach should be via so-called adaptive management policy and practice. This is essentially 'learning by doing' with policy and practice being constantly monitored and re-orientated/changed as experience is gained during implementation. Such an approach accepts the inherent complexities and uncertainties that often shroud the utilisation of marine resources (Turner, 2000).

Problems of resource overexploitation and/or environmental quality degradation tend to have multiple causes and are evolutionary. Complexity and the power of networks (natural and human behavioural) serve to make management and decision making tasks very onerous with potentially very costly consequences when the wrong measures are introduced e.g. the recent worldwide financial crisis (Krugman, 2009).

We are of the opinion that decision makers should stop asking for 'easy, cheap and simple to understand' environmental indicators and accept the complex reality that is our environment. Given the panarchic character of natural systems realistic base environmental indicators should be anchored to a thorough examination of the functioning and the structure of ecosystems (de Jonge, 2007; de Jonge et al., 2003, 2006) instead of selecting for the 'cute and cuddling' icons of any ecosystem without knowing what they ecologically represent.

An adaptive management process should be composed of a number of sequential but overlapping components:

* baseline science and indicators to inform in terms of the ecosystem structure, process and forcing vectors that condition the coevolving ecological and socio-economic marine system and its inherent trends;
* the application of methods and techniques (the tool box) for the assessment of the marine system's status and future prospects;
* focused analysis of contemporary 'key' and potentially significant emerging issues due to overarching environmental change;
* participatory and deliberative methods and techniques to foster social dialogue amongst all relevant interest groups, and to search for 'values' consensus/majority positions;
* modelling to compare alternative policy option outcomes;
* further development of appropriate indicators and adequate monitoring and review procedures.

1.2 Present EC Directive-related failures in marine management

The implementation of the WFD (EC, 2000) is based on monitoring selected parts of the ecosystem. The present focus is on phytoplankton, macro-algae, angiosperms, benthic invertebrate fauna and fish while the most important carbon fluxes in ecosystems are at the level of microbes, detritus and primary producers. Baird et al. (2004) conclude that about 99% of the recycling involves only some compartments with mainly sediment bacteria and particulate organic carbon (POC) as detritus. Something comparable is also observed for a part of the Schelde estuarine system (van Oevelen et al., 2006). The latter concluded that the herbivorous and detrital-microbial pathways function highly autonomously. How important the detritus related pathways are as stabilizers of the ecosystem functioning, however, needs further research. From a holistic ecosystem perspective as well as the EU 'ecosystem approach' the present implementation of e.g. the WFD directive seems still highly sectoral in its approach. The accepted low minimal sampling frequency further supports the conclusion that the collected data are not particularly helpful for analysis related to coastal policy making and management (de Jonge et al., 2006).

The required WFD related river basin management plans are focussing on rather general elements and not on anomalies in the system functioning, structure or condition in relation to created human pressures.

The other relevant strategy, the Marine Strategy (EC, 2002) leading to the adoption of the Marine Strategy Framework Directive in 2008 (EC, 2008) aims at integrating the practices from other relevant EC Directives in an 'ecosystem approach' which is then considered to be the EU Strategy for Sustainable Development. This strategy is in its implementation phase now. Although this strategy is much more ambitious than the WFD it is not providing a clear technical strategy with supporting instruments or tools, but is based on fourteen objectives (EC, 2002). Some relevant objectives are to: protect nature, stop habitat destruction, change fisheries management, improve water quality at all levels and from all sources, eliminate litter, (more recently also to reduce noise), reach a more effective co-ordination and cooperation, pursue the new strategy at the global level and finally (objective 14) to improve the knowledge base on which marine protection policy is based. To us this forms the challenging basis and justification for providing some direction to the creation of technical tools which can be integrated to more effectively monitor the developments of the formulated EU aims.

1.3 Aim of paper

The paper's aim is to present an overview of an overarching framework and component instruments or tools that can be combined or integrated to arrive at a set of suitable indicators to judge the systems condition or status in terms of health, resilience, carrying capacity and related aspects. We will conclude by giving direction to 'how' to move forward in the spirit of the Strategy Directives.

2. THE INTEGRAL SYSTEM

2.1 Ecosystem research and the EU ecosystem approach

There is in ecology a long historical acceptance of the importance of studying systems entirely or holistically and not partially. System theoretical philosophies (Bertalanffy, 1968) and the ecological work of Odum (1971) have set the scene for a general view that studying the total energy flows through any ecosystem in sufficient detail could create a rational basis for understanding the complex functioning of food webs, if not yet the role of its complex species structure. Published ideas on the need for detailed integrative research of preferably unfragmented systems date back to the 1970s (Baretta and Ruardij, 1988; Ehrlich and Goodland, 1987; Holling, 1978; Kremen et al., 1994; Mooney, 1983; Ulanowicz, 1980).

Despite its importance every scientist realizes that this sort of ecosystem research is expensive because it can only be executed by relatively large teams of specialists. This in itself has been demonstrated to be enough reason for not getting it supported by governments or by their regulatory agencies. A very successful European example was a Dutch research team (Biological Research Ems-Dollard Estuary) which, based on the ecological ideas of Odum, investigated the main energy fluxes of the ecosystem in the Ems estuary over the period 1972 - 1985. The relations were quantified from bacteria and detritus up to the fish, while covering anoxic as well as oxygenated conditions in water and sediment and including water chemistry and physics (Anonymous, 1985). The final result was one of the first successful mathematical computer simulation models (Baretta and Ruardij, 1988). This success was only possible because of a clear vision in combination with one clear goal (integrating the collected data in one system i.e. a computer simulation model) and leadership (scientifically keeping course and continuously convincing impatient politicians of the value of this type of research).
Despite its success the Dutch government was not willing to further fund this strategic 'know how' research (creation of a knowledge agenda) and also not to further develop the modelling of Dutch coastal ecosystems on that basis. Consequently the integrative BOEDE research stopped in 1985 after which the scientists decided to report to the scientific community (Baretta and Ruardij, 1988). Although not scientifically rational, this political attitude is 'understandable' given the sectoral/local pressures that can arise as the rate and extent of environmental change increases. Long run strategic decision making is often much harder to take than following the line of the more often locally popular 'soft decision making' e.g. protecting fishing communities etc.
The progress in ecosystem research made since the late 1980s in, for instance, The Netherlands is fragmented because the field research carried out since then was usually not part of a master plan. Nor was the plan linked to an integrated ecosystem study concept founded on the 'ecosystem approach'. Rather it was either part of progress in fundamental research (development of concepts), or part of 'problem oriented' research (sectoral problem solving). Problem oriented research implies that the scientific task is narrowed to solving a specific problem like the effects of shell fishery on birds, the effect of gas drilling on bottom subsidence, the exploitation of the large scale offshore harbour development near Rotterdam effect on the large scale transport of fish larvae and suspended mud in the North Sea.
Contrary to the intentions of the Marine Strategy Framework Directive (EC, 2008), this sort of research does not analyse the functioning of entire ecosystems to determine its 'condition', 'carrying capacity', 'health' or 'resilience', or to contribute to the development of any

knowledge system or decision making instrument which is currently not available for the Dutch government and their agencies. The same is true for most other EU member states. More leadership, more vision and less politics at the national governmental level (holding for politicians as well as their senior advisors) could have resulted in a more beneficial development of the knowledge relevant to integrally manage our coasts and estuaries (de Jonge et al., 2006).

Crucial here is to define what to investigate what to monitor and how this then can be related to the relevant social and economic parts of the integral system to support sustainability in line with the Brundtland Commission's view (strong sustainability).

2.2 Socio-economic research in relation to the EU ecosystem approach and the ecosystem services approach

The ecosystem approach (EA) and ecosystem services concept (ES) aim to provide an overview and analysis of the wider issues ('to look out of the discipline box') and to understand the functioning of the wider system encompassing the complex combination of the societal needs, economic market wants while underpinning the ecosystem structure and functioning. This 'big picture' contributes to a wider integrative and systematic perspective of ecosystems and then may help in setting more effective even incisive management actions upon specific local, regional national and international problems. Although efforts have been made to integrate all available SD components, applications of the EA have to date still remained strongly focused on ecosystem structure and thus biodiversity centred. In addition to biological structures we need an equally strong focus on ecological processes (at the proper scale) and to the environmental services benefits ('goods and services') in relation to human welfare. To understand better the dynamics of system change we also need to incorporate analysis of human pressures and drivers (sometimes in the form of future scenarios) as well as the natural stressors as indicated in Fig. 1.

While the EA thus provides a useful conceptual framework it is still too theoretical to be directly applied to managing the integral system. To overcome this drawback, efforts have been made to turn this EA concept into a more feasible and operational tool that allows us to study 'how' to judge the effects of humans on nature and how to direct future developments when considering the co-evolution of social human systems and natural systems. The ES concept helps to make this analytical transition.

2.2.1 Societal choice

The ES perspective requires us to review what socio-economic information is necessary to support EA/ES implementation. First socio-economic information (on environmental drivers, pressures and changes) from the local up to the regional and international scale that is relevant at the ecosystem level should be included.

Aquatic ecosystems, and more specifically estuaries, are considered to belong to the most productive and valuable ecosystems around the world (Costanza et al., 1997; Jørgensen, 2010). The increasing population densities and subsequent increasing socio-economic demands (exploitation and modification of these systems) lead to increased human stress on these systems. Despite all the stress, ecosystems still have the ability to provide a wide range of ecosystems services benefits, such as food production and recreation, while at the same time providing a wide panoply of regulatory and support ecosystem services, as nutrient cycling and water purification (Balmford et al., 2002, 2011; Bateman et al., 2011; MEA, 2003; Turner et al., 2003).

The socio-economic research needed to support an EA/ES implementation should be such that we get an output which is directly useable in the decision making process. We have several conceptual approaches at our disposal (e.g. DPSIR, Ecological Sustainability Trigon (EST) and social-ecological systems (SES) for integrating qualitatively and/or quantitatively the interactions between the ecosystem and the socio-economic system (Berkes and Folke, 1998; Brock et al., 2009; Costanza et al., 2001; Marques et al., 2009; McLusky and Elliott, 2004; Turner et al., 1998).

This sort of analyses involves a clear definition of the main activities, stakeholders involved and general society characteristics (e.g. demographic data), monitoring of the stocks/flows from and to the system (‘input-output analysis’), the degree of human dependence on it, and the main impact it has on both the ecosystem and the human population. Therefore, an insightful characterization of the different forms of capital (natural, human, manufactured, and social; Costanza, 2000) of an ecosystem has to be performed.

Apart from the requirements defined above, the EA/ES also offers opportunities for the socio-economic disciplines to bridge the gap so that EA can play the role in ‘sustainability’ as foreseen by the European Union Commission. The term ‘sustainability’ now occupies a prominent position in the political lexicon and political agendas from the local (e.g. regulations controlling pollution sources) to the international levels (e.g. directives controlling water quality by WFD and protecting biodiversity by the Convention on Biological Diversity), reflecting the growing wider societal level of concern (Costanza, 2000; Duit and Galaz, 2008; de Jonge, 2007; Marques et al., 2009).

2.2.2 Defining environmental limits

By combining and substituting between the different forms of capital (physical, human, social and natural) the wealth creation process has expanded enormously (albeit unequally on a global basis). A big issue that now faces contemporary society is how much further can natural capital be substituted for via technological advances before thresholds are breached and unexpected system change occurs, possibly signalling unsustainable levels of ecosystem utilisation. If we adopt a definition of sustainability that implies that the current human generation must pass on a stock of capital to the next generation that is no less than it is now, we can distinguish two views about the conditions necessary to realise sustainability- the weak and the strong sustainability positions. The former view maintains that sustainable development can be achieved by transferring an aggregate capital stock value to the next generation that is no less than the current level. It is based on an optimistic assumption of the power of technological innovation and the continued substitutability of natural for other forms of capital. The strong sustainability view does not accept the indefinite substitution possibilities axiom and focuses on the existence of ‘critical’ natural capital (e.g. life support systems, the hydrological cycle, etc) that cannot be substituted for, either literally or on cost grounds. In reality there are a number of ‘middle ground’ possibilities.

The acceptance of a stronger or weaker version of the sustainability worldview, however, does have implications for ecosystem management and the further development of environmental decision making. The use of economic cost benefit analysis (ECBA) as a decision making support system implies a decision rule which selects options that maximise individual human welfare measured in monetary terms. So the monetary benefits for example of utilising ecosystem services in some way can be compared with the costs of that option. The closer we move towards the adoption of a strong sustainability position the lesser is the scope for CBA application, because the scope for natural capital substitution is assumed to be less. Instead we must substitute rules such as the precautionary principle which prioritise conservation of ecosystems. Most recently it has been argued that the natural capital stock and flow approach to environmental management should not serve to obscure the equally pressing need for radical reforms of institutions and governance (Norgaard et al., 2009). Much depends

on how pressing the global sustainability constraints really are, or what are attitude to collective risk taking should be. But institutional and governance issues are clearly key parameters that need to be addressed in any serious sustainability dialogue and so far progress at the national and international level in this dimension has been limited.
Ideally ecosystems would be managed under sustainability rules, in practice there are a number of acknowledged reasons why ecosystem degradation continues in some contexts unabated. These reasons include both market failure and poor governance. Taking the former, markets fail to allocate ecosystem resources efficiently because of lack of information on ecosystem functioning, and ecosystem service (benefit) prices and non-market values (see e.g. de Jonge et al., 2006).

Only some ecosystem services are traded in markets and even then the market prices may not reflect the total economic value of that particular asset. Often the full environmental costs (externalities) of the economic activities involved in utilising ecosystems are not reflected in these prices (Barbier et al., 2009; Perrings et al., 2009). According to Daly and Farley (2004), most of the services and goods provided by natural systems do not gather all the characteristics required for an efficient allocation in markets (excludability, where property rights are included, and rivalness). Therefore, effective policies that characterize a specific good or service should be applied to the specific combination of excludability and rivalness if optimal allocations are aimed at (Daly and Farley, 2004). It has therefore been argued that appropriately assigning market and non-market values to environmental assets is important for environmental management. A range of methods and techniques have been devised to assign monetary values to ecosystem services in the absence of market price data including survey based methods (e.g. by contingent valuation studies and choice experiments as reported by Barbier et al., 2009 and Mitchell and Carson, 1989). In the economic literature a number of issues have been identified which serve to complicate and limit the application of the economic valuation of ecosystem services. They include the spatially explicit nature of some ecosystem services provision; the requirement that ECBA must be based on so called ‘marginal’ changes in service provision and not total system collapse/loss; the avoidance of double counting of benefit values; and the complications caused by non-linearities in benefits provision and threshold change effects (for more detail see section 3.4 in this paper and Bateman et al., 2011; Fisher et al., 2009). Studies have also shown the limited contributions of it, given a non-exhaustive data set (e.g. Pinto et al., 2010). A further more accurate valuation of biodiversity assets is required but this implies an enormous task with an uncertain outcome.

This fact has lead to a gradual shift from further developing biodiversity based indicators to suggestions of looking for new ‘paradigms’ (de Jonge et al., 2003) and for monitoring and assessing the pressures and resulting state changes within the ecosystem (Levrel et al., 2010) or even the integral system (de Jonge, 2007; de Jonge et al., 2006).

2.2.3 Conservation of ecosystem structure and functioning to maintain ecosystem services

There are various options possible to maintain the flow of ecosystem services (ES). A very conservative and safe one is to conserve the balance between a specified level of biodiversity and the functioning of the system. This is nearly a ‘contradictio in terminis’ because it supposes that we are able to define and to judge this balance which is currently not the case. We do not know how to guarantee and maintain a particular stock and related flows under naturally varying conditions as they occur in natural and open systems to guarantee a particular level of ES. This also means that we need to fully describe in sufficient detail the relation between the ecosystem structure (biodiversity) and functioning as a solid basis for human well-being estimation (e.g. Naeem et al., 2009).

2.2.4 Level of action at different relevant temporal and spatial scales

A further question is what temporal and spatial scales are relevant when talking about ecosystem services in relation to sustaining human life in general. Through the integration of the ecosystem’s inherent processes, the associated biodiversity and its sustainable use, the ecosystem approach focuses on conserving natural systems for their inherent value and for human well-being (de Groot et al., 2010; Nunes and van den Bergh, 2001; Vitousek et al., 1997). In an overall perspective, socio-economic research has, when applied to the EA framework, the capacity to (Sutinen, 2007) analyze and explain the spatial and temporal variations in the uses of the principal ecosystem resources; assess the market and non-market value of human uses of natural services of ecosystems; assess the benefits and costs of protecting and/or restoring ecosystem resources; and assess the socio-cultural values of the uses of ecosystem resources and services. The use of comprehensive approaches (e.g. ES inventories in EA studies) to evaluate significant ecological, social and economic costs and benefits facilitates the work of decision makers regarding the implementation of management and conservation strategies (Pinto et al., 2010). Two scale-related problems are encountered when assessing ecosystem services (Heal and Kristrom, 2005): (i) the scale at which certain functions become important is not always the same and (ii) problems may arise when integrating and aggregating information at multiple scales where interrelations and feedback loops may operate at scales above the level being assessed. According to Limburg et al. (2002), scaling rules that try to describe the provision and delivery of ecosystem services have yet to be quantified and defined. Moreover, issues such as cumulative pressures and intricate interrelations among factors, internal and external to the system, are also determinant subjects to be considered when looking for optimal allocation and management of ecosystems.

2.3 Coupling the social and ecological part of the integral system

The presence and activity of humans has globally dramatically changed the environment. Given the complex behaviour of the human community and the ecological system (Levin, 1999) and thus also the connections and interactions between them as part of the 'integral system' (Fig. 1) a relevant question here is how we can relate human activity to environmental response in a way beneficial to management. This question is far from trivial because conventional, mainly sectoral, approaches are of limited use (see above and e.g. Holling and Meffe, 1996). A well known example is that of setting quotas for the fishing industry. This has been shown to be not enough regulatory effect to manage fish stocks because of the complex responses and feed backs within the ecological food web. This limitation is mainly due to the fact that the species structure is not static but dynamic in terms of composition and abundance. Species respond to changing environmental conditions as well as changes within the species structure and abundance itself (see above). All systems thus show different structure expressions or representations and also different qualities in space and time (Fig. 2). At another abstraction level these differences may be less pronounced as for instance is the case when considering the 'functioning' of the structure under consideration. At that level many systems are more comparable with each other (Baird and Ulanowicz, 1993; Herman et al., 2000; de Jonge et al., 1995; van Oevelen et al., 2006). Thus, despite the available panarchy in structures the functioning of it may be more or less the same.

There are several ideas on how the interaction between the social and ecological systems can be realized and how the systems condition or health could be judged. We are not going to contribute yet more ideas to what is available at this stage but will explore from what is available and applicable for the desired integration and judgement. Moreover, we will describe what is going on in terms of relevant system wide research and monitoring programs and how this all does or does not fit our ideas. Apart from that, we will further (see the figures in this paper) visualize how the socio-economic and ecological systems could technically be connected to each other. This point is of utmost importance since concepts more than problems should lead to how and when to monitor what part of a specific system.

The application of the ecosystem approach would allow the integration of ecological sustainability, economic efficiency and social fairness into a concise framework. Marques et al. (2009) provide a set of scenarios for alternative options of managing systems, considering the social conditions, ecological status and services provision spheres. To guarantee that accurate decisions are undertaken, a clear perception of the society's goals, at both short- and long-term, must be defined (Brock et al., 2009; Costanza et al., 2001). Thus, when choices have to be made between the ecosystems' conservation and the expansion or maintenance of human activities, a comprehensive knowledge of the impacts and importance that these activities may have on the natural environment and on the services provision have to be taken into account.

3. Tools to guide management actions

3.1 Conceptual assessment design

A first step in designing an ecosystem approach functionality is the clear definition of the ecosystem properties, problems and goals to achieve.

An effective sustainability assessment method should provide overall information without loss of information of system parts. The approach must, therefore, be necessarily holistic covering the variation in system aspects like e.g. performance, viability, carrying capacity and resilience with the largest impact on the overall system's sustainability (Bossel, 2000). This implies that information must be put together in an integrative and cumulative way and using a method or instrument that recognizes all the relevant system components, its values as well as the expected future values of the ecosystem and the social system and based on current and expected future desired human activities.

Goals can be articulated to express the current trends and provide the basis for the entire assessment (Hardi and Zdan, 1997), including the selection of the key orientors to be followed. According to Bossel (1992) *'orientors are aspects, notions, properties or dimensions which can be used as criteria to describe and evaluate the system's developmental stage'*. An orientor is, therefore, built or composed by a set of sectoral indicators. To answer the need for suitable communication, one-measure sectoral indicators and composite indicators are increasingly popular for policy makers (compare the well-known economic indices). They are also considered as useful for public involvement in conveying information on systems performance, considering environment, economy, society, or technological development (Singh et al., 2009). These indicators, however, should be derived from state of the art research and surveys.

In general, ecosystem approach studies require integrative tools that reveal the system status, and further demand a framework application that will work as a road-map to be followed (Knoflacher et al., 2003). This brings us to the point of what we have available now or require in the future.

3.2 Single and composite indicators and tools

3.2.1 Biological indicators

From an operational point of view indicators have in general to fit in a well-accepted sequence of objectives, monitoring programs and management measures (McLusky and Elliott, 2004). In order to be considered as a ‘good indicator’ for ecosystem conditions, several requirements must be fulfilled. For example Salas (2002) considers that a good ecological indicator should be: (1) easy to handle, (2) sensitive to small variations of environmental stresses, (3) independent of reference conditions, (4) applicable to extensive geographical areas, (5) relevant for policy and management needs. Several schemes and classifications of catalogued indicators are available (see, for example, Belfiore et al., 2003; Dale and Beyeler, 2001; Hellawell, 1986; Marques et al., 2009). Hence, it emerges that as long as indicators fulfil the requirements under the heading ‘characteristics’ they may vary from species, via processes, values of boundary conditions to resource concentrations. The difficult task is to derive an indicator or set of indicators that together are able to meet these criteria. In fact, despite the panoply of ecological indicators that can be found in the literature, very often they are more or less specific for a given kind of stress, applicable to a particular type of community or site-specific (Pinto et al., 2009; Salas et al., 2006). Moreover, another big problem is that the system functioning or the species structure ‘story’ related to or behind the measured values is usually not clear. For that reason biologists and ecologists still look for and contribute to the development of new indicators.

The conceptual idea behind the development of biological indicators based on a single species or species assemblages is that they are supposed to reflect the effect of any stress or complex of stress put to the system (Pearson and Rosenberg, 1978; Rosenberg et al., 2004).
For open and dynamic shallow coastal systems it is difficult to accept that one single species could be used to indicate the systems quality or condition (see also Elliott and Quintino, 2007). Instead composite biological indicators based on benthic macrofauna assemblages have been favoured because these benthic animals live relatively long and thus may have incorporated within the species assemblages the negative effects of the system's stress. The species composition then thus depicts the effects of the total environmental stress (cf. Pearson and Rosenberg, 1978). At the species structure level the other extreme is the use of a multispecies indicator (AMOEBA approach; ten Brink et al., 1991) where the relative abundance of circa 30 species are plotted, and in a radar plot in an attempt to view holistically the system's quality. However, from an ecological point of view a list of species and its (relative) abundance is not enough to qualify the functioning of that system. This is caused by the observed strong inter-annual variation in species abundance in shallow coastal ecosystems (de Jonge, 2007). Moreover, according to Elliott and Quintino (2007) one of the main causes for strong variations in coastal and estuarine systems is often natural factors instead of human activities.
Recently several authors have tested the among agreement of a large number of biological indicators. Blanchet et al. (2008) investigated the AMBI (AZTI marine biotic index; Borja et al., 2000), BENTIX (BENTIX biotic index; Simboura and Zenetos, 2002), Shannon–Wiener diversity, BQI (Benthic Quality Index; Rosenberg et al., 2004) and BOPA (Benthic Opportunistic Polychaeta Amphipoda index; Dauvin and Ruellet, 2007) biological indices in two semi-enclosed sheltered coastal ecosystems along the western coast of France. Chainho et al. (2007) studied subtidal assemblages of the Mondego estuary focussing on the application of the Margalef, Shannon–Wiener, AMBI and W-Statistic indices. Patrício et al. (2009) studies three areas within the Mondego estuary by the Margalef, Shannon–Wiener, Berger–Parker, taxonomic distinctness measures, AZTI marine biotic index (AMBI), infaunal trophic index (ITI) and eco-exergy based indices. The results (Blanchet et al., 2008; Chainho et al., 2007; Patrício et al., 2009) were disappointing in that the agreement between these indicators was either absent or weak. Patrício et al. (2009) conclude that presumably the developed indicators describe different aspects of the biological quality. Based on the above, and in agreement with de Jonge (2007), we arrive at the conclusion that attempts should be made to integrate species abundancy with aspects representing the systems functioning.

3.2.2 Determining habitats and its ecological characterisation

Physical factors like salinity, sediment composition, temperature, tidal range, elevation of intertidal stations can be used to define (potential) habitats. These then can be visualized in a GIS (Geographical Information Systems) application. The principle is exemplified in Fig. 3 and shows that based on only the 3 factors (elevation, current velocity and the distinction between saline and brackish) 8 different potential habitat classes can be distinguished. Because of the distinction between brackish and saline we even end up with potentially 14 different habitats (6 in the brackish zone and 7 in the saline zone). An example of the scale and distribution of these potential habitats is also presented in Fig. 3 for the Schelde estuary (The Netherlands). In combination with the probability of occurrence of species as function of the same type of factors as described here one can obtain a picture of the potential or possible species assemblages within the different zones as depicted in Fig. 3 (see Ysebaert et al., 2002). An impressive documentation of the necessary analyses can be found in Ysebaert and Herman (2002). In our opinion this technique can be used to characterise relatively large units within an estuarine or coastal system. When these potential habitats (or ecotopes) have been described in terms of species structures (communities) then it may offer the possibility to narrow focus to this aspect instead of all the details related to these 'habitats' or 'suitability maps' (HABIMAP; de Jong, 2000a) for the occurrence of specific species assemblages. Of course there are many problems related to this approach (see Ysebaert and Herman, 2002) but also challenges as also shown by Thrush et al. (2003). Therefore, we have decided to look at the potential opportunities to be able to make a next step which is also to incorporate the rest of the ecosystem and to apply the approach for management purposes.

[insert Fig. 3 here]

3.2.3 Dynamic modelling

Models range from mathematical and statistical models to functional models and from descriptive or phenomenological models to causal or deterministic models. Despite the fact that there is a lot of criticism in using dynamic models for predictive purposes (e.g. Haag and Kaupenjohann, 2001) they are, stimulated by requests from governmental authorities, widespread in use. In an interesting and convincing article Haag and Kaupenjohann (2001) explain that due to the dynamical system paradigm these systems are conceptually and necessarily closed systems requiring a fixed set of ‘a priori’ defined parameters. They further explain that ecosystems are conceived as conceptually open, self-modifying systems, which itself produce novelty and new parameters and which cannot be severed from their environment while the dynamic models cannot escape their own constraints. Thus the predictive capacity of these model systems is not at all warranted so that they have to be considered as deficient instruments in reducing the uncertainty as to future system behaviour. Modelling exercises for decision-making need to take into account the transparency of the process in order to facilitate the participation of stakeholders. When modern concepts such as self organisation are applied and system structures can develop freely then the question about the uncertainty of the capacity to predict arises as well. However, given their magic irradiation to decision makers and managers it is not realistic to assume that they will disappear as decision supporting instruments within a short period of time. Therefore we are the opinion that it is better to look for possibilities to use these instruments in a slightly different way which is to and in combination with other applications.

3.2.4 Ecological Network Analysis

A static approach, with potential for dynamic use, and which can be combined well with dynamic modelling is ‘ecological network analysis’ (ENA) (Ulanowicz, 1980, 1986, 1997). Mathematically ENA is built on the basics of graph theory and matrix algebra. As tool ENA is also based on the thermodynamic laws, Lindeman’s trophic analysis (Lindeman, 1942), the Finn cycling index (Finn, 1976) and the input–output analysis (Leontief, 1951). One of the first who applied part of this approach, the input-output analysis techniques, to ecosystems was Hannon (1973). The ENA analysis requires a ‘quantified’ food web because it is species (or functional species groups) oriented (see example of the food web in Fig. 4). In its simplest form it is a network consisting of nodes tied to each other by arrows. ENA may be a helpful tool in judging the systems condition by an available set of quantitative system indicators. The data needed for ENA are the same as needed for dynamic modelling exercises and being biomass (B), physiological requirements (P/B), loss terms (respiration or dissipation, export, catch) and relationships between compartments (diet relationships: who eats what, whom and how much). Information needed for the ENA food web analysis is the food web structure and compartment related data so that a number of indices can be calculated. The ENA approach seems to be a good candidate for further application in both analysing the functioning and judging the quality of ecosystems.

[insert Fig. 4 here]

ENA is not a tool in itself but has been further developed by combining techniques used in the social sciences (see for an example Luczkovitch et al., 2003). New techniques become increasingly important in especially studies related to social aspects in all sciences (e.g. Janssen et al., 2006; Martínez-López et al., 2009) but until recently less so in economic studies (see also below). A short and clear overview of the origin and potential of ENA is published by Fath (2004). Recently Patrício et al. (2004) describe in detail the procedural steps for the application of ENA. Energy budgets can be developed using ‘‘ecopath with ecosim’’ modelling software which can be found at (http://www.ecopath.org). This results in balanced budgets for each trophic group. Values on consumption, production, respiration and ingestion by Ecopath with Ecosim are subsequently imported into (http://www.glerl.noaa.gov/EcoNetwrk/ or http://www.cbl.umces.edu/~ulan/ntwk/network.html; Ulanowicz, 1999) to calculate annual biomass budgets for each compartment. The structures of trophic levels and cycling as well as the input-output analysis for a given network can then be analysed and the system properties be calculated using algorithms described by Ulanowicz (1986).
The trophic status can be assessed by the trophic analysis which calculates trophic efficiencies among different estuarine systems according to a standard straight-chain network (Baird et al., 1991). The trophic efficiency between any two levels is defined as the amount a given level passes on to the next one, divided by how much it received from the previous level (Ulanowicz and Wulff, 1991). The energy flow networks can be visualized in a canonical trophic form (‘‘Lindeman spine’’; Ulanowicz, 1997). Connectance indices are estimates of the effective number of links both into and out of each compartment. The Finn Cycling Index (FCI) quantifies the proportion of total system throughput (TST) that is devoted to the recycling of carbon (Finn, 1976). For more indices the reader is referred to Ulanowicz (1980, 1986, 1997). Finally, the input-output analysis quantifies direct and indirect trophic effects for each component in the network.

[insert Fig. 5 here]

Apart from the main analyses, there are a number of ecosystem indices which can also be used for practical purposes. Some of the main indices (cf. also details in Fig. 4 and 5) are briefly described. Total system throughput (TST) is the overall activity of the system and which is given by the total sum of all the transfer processes in that system. Ascendency (A) in Fig. 5 indicates the organisation of the flows and the magnitude of them. It is interpreted by Ulanowicz as “the tightness of the constraints channelling trophic linkages”. A higher ascendancy indicates a food web with stronger cycling due to ‘trophic specialists’ and/or higher efficiency while lower values indicate a more generalist-based system with consequently lower transfer efficiency and decreased cycling. It also represents the degree of organisation (‘developmental status’). Average mutual information (AMI) in Fig. 5 is then the unscaled form of the ascendency (A) and measures the average amount of constraint exerted upon an arbitrary quantum of currency as it is channelled from any one compartment to the next. Developmental capacity (DC) in Fig. 5 represents the diversity of the system flows scaled by the total system throughput (TST). It thus also is an index for the systems complexity. Overhead (O) in Fig. 5 is an entropy term and a measure of inefficiency of the material (carbon) flow through the food web. It is a ‘disorder’ term caused by the system ‘dissipation’ (e.g. respiration), the ‘redundancy’ of relations between species compartments and the ‘export’ from the system. It is the amount by which the capacity of a non-isolated system exceeds the ascendency. In terms of the flows it resembles the redundancy but including the transfers with the external world. Redundancy (R) quantifies the degree to which pathways parallel each other in a network. The fluxes between the different trophic levels, which form the basis for the indices, are given in the strongly simplified food web of Fig. 6.

[insert Fig. 6 here]

The above indicates the complexity within ecosystems and also demonstrates that straight forward description of ecosystems by only species assemblages (e.g. AMOEBA approach; ten Brink et al., 1991) cannot be useful to judge the condition of the system. The system’s structure as well as the system’s functioning need explicitly or implicitly to be incorporated in any indicator. However, the situation is even more complex than described so far. The existence and the importance of high internal ‘connectedness’ compared to the connectedness between systems is also an aspect already mentioned by many others (e.g. Jörgensen and Müller, 2000). Based on more general ecological considerations these authors mention that ecosystems are not only emergent in their expression and show cycling of material but also show self-regulation and self-organisation based on feedback loops. We possibly may be able to incorporate this sort of dynamics when calculating developments at the habitat levels where we are dealing with a restricted number of parameters. Within the context of the present paper, application to the ecosystem level may yet be a bridge too far.

3.3 Integral indicators and tools

Coupling of data from very diverse fields as ecology, economy and social spheres requires a framework for guiding the integration. One approach could be the use of the ‘integral system’ as starting point to fill the gaps among information and data (Fig. 7). The social-ecological systems (SES; as proposed by Berkes and Folke, 1998) connection, represented in the above diagram by the ‘final services’ box, assumes that a series of concepts, such as resilience, complexity or sustainability, are inherent to this kind of analysis, once they will impact or even determine the flux intensity among compartments. The dynamic links are represented by the DPSIR related steps (driver, pressure, state, impact and response) (see section 3.5).

[insert Fig. 7 here]

3.3.1 Resilience and carrying capacity as conceptual integral tools

Resilience may be defined as the capacity of any system (natural, agricultural, urban) to cope with external disturbances without shifting into a qualitatively different state (Gunderson and Holling, 2002). Many scientists and policy makers consider resilience as an important indicator of ecosystem integrity, allowing to determine critical thresholds and the minimum requirements for ecosystems functioning, and consequently to services provision. Lenton (1998) suggested that complex ecological networks act as an adaptive system by stabilising feedbacks and thus reducing system perturbations. The same is thought to be true for ecological succession which is assumed to lead to mature-stage ecosystems with good resistance to external perturbations (Odum, 1969). Based on these discussions Kristensen et al. (2003) suggested that human society resilience could also be considered as representing an ecological goal function.

The carrying capacity concept is intimately associated with the notion of thresholds and a certain optimum and maximum level in development of a system and its compartments. The system’s carrying capacity may be defined as the point where the biomass of a given population stops increasing (achieving the biomass maximum carrying capacity). This development (governed by resource limitation or scarcity in space) is considered as an ecosystem property (Dame and Prins, 1997). This definition of maximum level of carrying capacity may differ from the economic carrying capacity (Smaal et al., 1998) that is related to exploitation and usually underlies management strategies.

Both issues are complex and demand for integrative ways to measure it. Some efforts to measure the resilience and carrying capacity of a system have been carried out by e.g. Berkes et al. (2003); DeAngelis (1980); Kristensen et al. (2003); Ludwig et al. (1997); Smaal et al. (1998), but there is still no consensus regarding the relations between properties of resilience and carrying capacity and functions of ecosystems. Most of the attempts have been focused on model development and implementation. For example Kristensen et al. (2003) employed a model to estimate the role of resilience on some goal functions of systems. They concluded that the maximization of resilience leads to the optimization of other goal functions in the system (e.g. phyto- or zooplankton biomasses, nutrients flux). However, the authors highlighted the need to further analyse the mechanism underlying the maximization of the systems resilience. The same happens when models on carrying capacity are applied to estimate optimum growth and exploitation of commercial species (e.g. Duarte et al., 2003; Smaal et al., 1998), or even to evaluate the outcome of management strategies and promote efficient measures (e.g. Thébault et al., 2008).

Within the context of the present paper it is very difficult to apply ‘resilience’ technically because it is, so far, a definition that has not been reached a proper implementation level. Therefore, and alternatively, it may be better to apply ENA related indices to indicate the quality of the ecosystem and to use that as a basis for further judging the anthropogenic role in the functioning of the integral system.

Carrying capacity is more easily executable than resilience. The use of carrying capacity is therefore recommended here because it is connected to the relation between two factors and often can directly or indirectly be related to growth (population) and production (bivalves, fish, plants) of parts of the ecosystem which can be analysed by ENA.

3.3.2 Ecosystem complexity and sustainability as conceptual integral tools

Complexity of ecological systems is an ambiguous term and usually may be related to structural, functional, or physical aspects of ecosystems (Adami, 2002). Nine forms of complexity were identified by Jørgensen (1997) giving special focus to the fact that the complexity is wider than just the interactions among species and resources (see above). A couple of measures have been developed to provide an integrative indicator of a systems (role) complexity level. For example, Adami (2002) has developed mathematical equations to cover the physical complexity of ecosystems, arguing that the total complexity would have to be defined as the mutual entropy of all organisms, about each other and the world they live in. In another perspective, and as pointed out by Jørgensen (1997), the way an ecosystem responds to perturbations has been widely debated in terms of stability. The complexity of the regulation by feedback mechanisms has not received much attention. Costanza and Daly (1992) argue that ecosystem health of complex system, defined by six properties (homeostasis, absence of disease, diversity or complexity, stability or resilience, vigour or scope to growth, balance between system components), may be given by a general system health index: $HI=VOR$; where, V is system vigour, O the system organization, and R is the resilience index. However, other indices for ecosystem health and complexity focus on exergy, specific exergy, and buffer capacities (Jørgensen, 1997), which also fits in this ecosystem health definition (Jørgensen, 1997). These last indices (may) are also in use to measure the system maturity, as Dalsgaard and Oficial (1995) did using the ECOPATH model regarding agro-ecological systems.

The interactions among biodiversity assets, ecosystem processes and functioning (BEF), and the services provided by natural or production systems has been widely studied over the years (e.g. Duarte, 2000; Griffin et al., 2009; Naeem et al., 2009). However, their quantitative relations are still poorly understood (de Groot et al., 2010), and more efforts are needed to develop a 'full integrative link' among compartments.

3.4 Social and Economic Analysis within a Decision Support System.

The integrated approach we have in mind needs to formally encompass socio-economic analysis and is guided by the acceptance of a strong sustainability viewpoint. Ecosystems are seen as suppliers of a range of ecosystem services through which humans benefit in terms of welfare or well-being. The analysis tries where meaningful to place monetary values on the benefits provided by 'healthy' ecosystems. But it is recognised that some services are not suitable candidates for monetisation e.g. so-called cultural services such as among others heritage landscapes and seascapes. The approach is also limited to service values that are largely instrumental and therefore it does not explicitly include pure intrinsic value which some commentators claim for nature. In terms of the political economy of nature conservation what is being proposed here is that the inclusion of socio-economic analysis within the decision support system (DSS) serves to supplement scientific and ethical arguments in favour of environmental protection. The EU Marine Strategy Framework Directive (EC, 2008) for example explicitly calls for (Article 8.1c) ‘an economic and social analysis of the use of those waters and of the cost of degradation of the marine environment’

The possible relative changes in quality status and the human-related activities which serve to pressurize the marine environment can be modelled within the DSS we have advocated. An initial scoping stage could be based on the D-P-S-I-R framework (see next section) and the temporal scale of the environmental changes can be modelled via scenario analyses. While future uncertainty will always remain problematic, scenario analysis (typically based on a ‘business as usual’ or BAU baseline trend assessment against which a range of different future paths can be assessed) offers a way of coping with uncertainty and provides policy relevant information. The process of economic analysis can only take place after policy issues have been identified within given spatial and temporal scales and scenarios and evaluative criteria have been established. Underpinning the whole DSS is of course the existing scientific knowledge base i.e. what is and is not known about ecosystem structure, process and functioning.

Once agreed, the policy issues and scenarios that are identified by the scientific and policy communities provide the context within which the socio-economic assessment can be constructed. Note however that this is not a one-way process. Feedback should occur between all stages of the assessment process and deliberative arrangements should be made with stakeholders, since questions that are thrown up by the assessment can help to refine the policy issues and scenarios that are of most concern to relevant stakeholders/user groups. In general most problem situations involve competing uses for coastal/marine resources and are conditioned by the governance that is in place.

The resource system policy issues under investigation will be composed of a complex mixture of environmental and socio-political driving processes, consequent environmental state changes which then impact on the provision of ecosystem services and their effects on human welfare/well-being. The distribution of the welfare gains/losses in society, together with existing policy measures and networks will influence policy responses. The economic analysis (cost benefit analysis CBA and cost effectiveness analysis CEA) seeks to evaluate the social welfare gains/losses from an economic efficiency perspective, tempered by any distributional equity considerations, other precautionary environmental standards and regional economic constraints (most often focussed on local employment and economic multiplier impacts which can result in cultural and community losses or gains, e.g. closure or restrictions on fisheries). The main distinction between CBA and CEA is that the desired outcome(s) is determined a priori in CEA e.g. the achievement of a legally set water quality standard at least cost to society, but not in CBA.

Economic valuation is often undertaken in terms of ‘opportunity cost’. This means that the value of an ecosystem service (or a damage impact avoided) is assessed through the ‘trade-offs’ associated with obtaining or maintaining the service flow. In principle it may then be possible to compare all relevant options and look for the ‘highest value’ uses of the ecosystem. Marine and coastal ecosystem services and benefits can in a simplified way be linked to four environmental impacts or effect categories relevant for human welfare:

- direct and indirect productivity effects (use values);
- human health effects (use values);
- amenity effects (use values); and
- existence effects (non-use values) such as loss of marine biodiversity and/or cultural assets.

Different valuation techniques will be appropriate for each of the four categories but note again that the symbolic and cultural values assigned to some coastal/marine features and land/seascapes lie outside the monetary calculus and are conditioned by social preferences and norms arrived at, over time. Through various forms of information transmission, art, literature, film.

Productivity effects related to, for example, fisheries, aquaculture, recreation/tourism and indirectly to services like storm protection, erosion reduction, etc, can be valued using market prices linked to changes in the value of output or loss of earnings. The approach needs a production function which is derived often through the use of bio-economic models (e.g. fisheries).They can also be valued using surrogates such as e.g. property prices, land values, travel costs of recreation and damage costs avoided. Health effects are valued by cost of illness measures or survey-based methods. Amenity effects can be assigned values through travel costs, property values or survey methods such as contingent valuation, contingent ranking and choice experiments. This latter group all use questionnaires to elicit individuals’ willingness to pay or be compensated in monetary terms for gains/losses of services. Finally, existence or bequest (from generation to generation) values can only be derived if at all via surveys. Because it is not possible to value all ecosystem services in monetary terms the DSS should include so-called multi- criteria evaluation methods (MCA) which quantitatively or qualitatively encompass a range of social/deliberative and ecological conservation perspectives. MCA as a framework can incorporate the results of CBA/CEA and provides weighted and scaled rankings of different options (DETR, 2000; Janssen, 1994; Olson, 1996).

In the literature five issues have been identified as critical to the appropriate economic valuation of ecosystem services. **Spatial explicitness** is important in order to clarify the level of understanding (or ignorance) of underlying ecosystem structure, process and functioning. This contextual analysis must then include appropriate socio-economic, political and cultural parameters in order to properly identify ecosystem services supply and demand side beneficiaries. Ecosystem services are therefore context dependent in terms of their provision and their associated benefits and costs. If we take the example of coastal wetland and their supply of carbon sequestration/storage services, it turns out that the net effect of this service is conditioned by the simultaneous release of methane. But the spatial location of the wetland and in particular the prevailing salinity condition plays a significant role in the carbon storage to methane emission ratio and consequent global warming effect (Luisetti et al., 2010). It is anticipated that the incorporation of spatial factors in ecosystem valuation is likely to become easier and more commonplace as access to GIS software and expertise increases (Bateman et al., 2006).

Secondly, **marginality** is an important issue as economics requires that for the valuation of ecosystem services to be meaningful such analysis should be conducted 'at the margin'. This means focusing on relatively small, incremental changes rather than large state changing impacts. However, given the scientific uncertainties which shroud ecosystem functioning, it is often difficult to discern whether a given change is 'marginal' or not, and when thresholds are being approached or crossed. Knowledge of the drivers and pressures on the ecosystems under study, as well as understanding of how the system is changing or might change from its current state is crucial. This has been called the system's transition path (Fisher et al., 2009; Turner et al., 2003;). It is important to know if the transition path is 'stepped' as in the case of a coral reef system or shallow lake/lagoon, or it is 'relatively smooth' such as in species invasion into an area. By identifying the transition path we can force the analysis to consider losses or gains in service/economic value between two distinct states of the system.

Thirdly, **double counting** may be a problem where competing services are valued separately and the values aggregated; or where an intermediate service (in economic value terms) is first valued separately but also subsequently through its contribution to a final service benefit. Thus a coastal wetland may provide nutrient cycling capability which then leads to better water quality for recreation and amenity. The economic value involved is restricted to the recreation/amenity gain, excluding the nutrient cycling the value of which contributes to the final service benefit value (higher quality recreation/amenity experiences).

Fourthly, **non-linearities** in services provision complicate valuation and system management e.g. shallow phosphorous- limited lakes flip from one state to another with dramatic effects on some services. Further, non-linearities can mean that marginal benefits are not equally distributed e.g. the storm protection benefit of a unit increase in mangrove habitat area may not be constant for mangroves of all sizes due to non-linearities in wave attenuation. If a cost-benefit assessment assumed linearity but service provision is in fact non-linear, policy option outcomes may be unnecessarily polarised (Barbier et al., 2008).

Finally, **threshold effects** i.e. the point at which an ecosystem may change abruptly into an alternative steady state, are problematic for CBA. For marginal analysis to hold true, the next unit of change to be valued should not be capable of tipping the system over a functional threshold or ‘safe minimum standard’. Given the uncertainties we currently face identifying risk will require expert input from ecologists and other scientists, risk analysts and ethicists etc and will ultimately require ethical/political choices to be made and deliberatively agreed.

The notion of total economic value (TEV) provides an all encompassing measure of the economic value of an ecosystem service supply. It is important to note however that TEV is always less than total systems value. A minimum configuration of ecosystem structure and process is required before final services can be provided. The system therefore posses ‘extra’ value known as ‘glue’ or ‘primary’ value (Turner et al., 2003). Because there is uncertainty over what is or is not a sustainable ‘healthy’ functioning state. In many contexts a precautionary approach to management has much to recommend it.

TEV decomposes into use and non-use values but it does not include other kinds of values such as intrinsic values which are usually defined as values residing ‘in’ the asset and unrelated to human preferences or even human observation. Cultural/symbolic values which groups of people have assigned to landscapes etc are also outside TEV. Nevertheless, apart from the problem of making the notion of intrinsic value operational, it can be argued that some people’s willingness to pay for the conservation of an asset, independently of any use they make of it, is influenced by their own judgements about intrinsic value. This may show up especially in claims about species ‘rights to existence’ but also as a form of human altruism.

3.5 DPSIR as a framework for further tool development

The DPSIR approach was developed by OECD (1994) and soon followed by further application (Turner et al., 1998). Since then it has attracted wide attention from the EU Commission, managers and scientists mainly because of its practicability. DPSIR can be defined as an operational framework identifying ‘drivers’ of change which lead to individual ‘pressures’ causing a different system ‘state’ which consequently lead to ‘impacts’ on human welfare which then require a policy/management ‘response’. The approach is attractive because it can be used in a very general way as a scoping framework assessing causes, consequences and responses to changes caused by any stressor. Apart from coupling the effects of human activities to the ecosystem (Fig. 1), it can also integrate ecosystem services and societal benefits (Atkins et al., 2011) which in this paper are indicated as ‘intermediate services’ (delivered by the ecosystem) and ‘final services’ (societal benefits). This is illustrated in Fig. 7. The drivers, pressures, state change, impacts and responses can be visualized schematically, see Rogers and Greenaway (2005). Such a scheme can be used during discussions among stakeholders but does not necessarily provide detailed enough information on the magnitude and significance of the ‘state change’. For management purposes we additionally need more specific quantified information. A next step then may be that the indicated DPSIR-factors are quantified and put together in a model describing the cause – effect relationship (including all the known feed backs) between the ecological and the socio-economic system as defined in Fig. 1 and now further visualized in Fig. 7.

The impact of these changes to the socio-economic system can first be quantified in purely ecological terms but subsequently in terms of changes in ‘goods and services’, as also suggested by others like Atkins et al. (2011). These are, for example, changed amounts of available stocks, harvest rates or recreation/amenity gains/losses.

3.6 Developing the integration among ecological, economic and social aspects

As indicated above, graph theory related science is used in ecology and widely in some social sciences, but less so in conventional economics. We take up the challenge to stimulate further steps in this integration. We emphasise that a lot of processes within human society as well as in our environment are conditioned by surface area and scale. Scales in human and biological social sciences may vary from square millimetres (bacteria, protozoens) to over fifty thousand square kilometres representing ‘eco zones’ (de Jong, 2000a). The notion that processes are operational at quite different spatial scales may be used to determine the proper scale for integrating conceptually the ecosystem and its intermediate service role with the economic system, its pressures to the environment and the environmental final services to society. This integrated picture could then be further developed to serve policy making and management activities. In the next paragraph we set out how we think that the integration could be started.

Apart from the spatial scale issue, ENA should preferably be performed to the level of species so that we end up with a very detailed and complex food web structure within which all the ENA related characteristics and indices can be calculated and compared over time. Changes in system quality can then be assessed by analysing the generated inter-annual indices time series. This sort of analysis could be done for different system conditions: i.e. if data are available a natural (reference) situation and a recent (perturbed) one due to e.g. anomalies in freshwater flows or weather conditions or human activity related system stress. The differences between the two states and the transition path should tell us something about the impact of the stress acting upon the ecosystem.

ENA can also be applied to a food web consisting of groups of aggregated species thus close to what is represented in Fig. 4 and the inlay of Fig. 2 (Sylt-Rømø Bay food web as published by Baird et al., 2004). A food web consisting of mainly functional groups and some dominant species (e.g. beds of the blue mussel (*Mytilus edulis*) on intertidal flats) added to it may be more simple and easy to create and handle than a very detailed one, but it will also produce different results because of loss of information. This also means that if ENA is going to be used for management purposes then before the start of the required sampling or monitoring programs a choice has to be made on the most appropriate 'aggregation level'. Exactly the same holds for the application of dynamic simulation models.

A third possible aggregation level is that of the available habitats within the system. The above described habitat mapping (HABIMAP; de Jong, 2000b) approach showed that there is a reasonably good agreement between different estuarine habitats or zones and the composition of the benthic macrofauna assemblages resulting from regression models (Ysebaert et al., 2002). These results then can be applied in habitat models like HABIMAP. The habitats assessed by the HABIMAP approach can also be used to define zones that play a functional role in relation to ecosystem functioning (behaviour of bacteria, plants and animals) and human activities. Fig. 8 for instance presents some examples: grazing by cattle and roosting of birds on saltmarshes, catching fish in the main channels, shrimp fishery during high tide above sandy intertidal flats fringing the channels and gullies, resting of seals on high elevated sandy flats bordering deep channels, predation on the intertidal mud flats by birds, etc. Thus, the 'spatial scale' related to specific habitats needs to be appropriate in order to connect human activities and some characteristics of the estuarine and coastal environment.

[insert Fig. 8 here]

We suggest here that network analysis techniques related to, but different from, ENA may also be helpful in connecting the economic and social cultural fields to that of the ecosystem for static as well as dynamic assessments. An interesting example has recently been discussed by Johnson et al. (2009) who applied a simulation-based continuous-time Markov chain model (SIENA) to determine the seasonal changes in the Chesapeake Bay food web. Apart from visual inspection they applied a statistical assessment to analyse developments. The visualisation was realized by the graphic analysis tool NetDraw (Borgatti, 2002). For more details the reader is referred to Johnson et al. (2009). Their conclusion was that in a qualitative way this holistic approach was successful in describing the changes in the food web from a highly complex one in summer to one of much lower complexity in winter (highest ascendency). Another interesting example was produced by Luczkovich et al. (2003) and showed the role of nitrogen involved in the food production system of Norway, visualized using the regular equivalence (REGE) algorithm and (3-dimensional) multi-dimensional scaling (MDS) approach (see for further details Luczkovich et al., 2003). In the example he describes the flows of the N import (by sea catches and animal production) and the transfers through the social-ecological system culminating in what he calls the 'wholesale food production'. In that network relevant parts of the environment are clearly connected to the social network in a quantitative analysis. Janssen et al. (2006) reviewed the importance of network analysis as an interdisciplinary tool which in this case was the social-ecological system (SES), while McMahon et al. (2001) provide tips to ecologists and social scientists interested in the use of network analysis. They pointed to the development of structural models to analyze human interactions (Wasserman and Faust, 1994) and specifically the conceptualization and testing of the interactions within complex systems as social and ecological systems. They further recommend the reduction of elements (nodes) and interactions (arrows) in such a way that the system becomes 'simple' enough to be analysed but still complex enough to reflect reality. Their main conclusion was "*that on many scales, social scientists, biologists, and physicists are all studying the same phenomenon. Most of the difficult problems modern society faces arrive in the form of complex structures such as economies, ecosystems, and societies"*. They argued for further and deeper interdisciplinary collaboration.

3.7 Challenges to the future

3.7.1 Possible actions

The DPSIR approach should be applied to the food web at different aggregation levels (detailed food web, food web based on functional species groups and based on the HABIMAP habitat units). A suggestion on 'how' to realize this is given in Fig. 9. This then should lead to insight into the sensitivity and thus also the practical usefulness of the different ENA indicators. It also results in the further quantification of input fluxes under the influence of the social system to the ecosystem, internal transfer, turn-over and export fluxes to human society (like catches of fish and bivalves; Fig. 9). As explained above, there are also stresses to the system related to the use of nature as a vehicle to realize a particular 'service'. The collection of food (fisheries) results for example in changes in the structure of faunal populations due to primary catch and by-catch plus accompanied possible habitat destruction. The use of nature as a vehicle may also lead to a certain system stress (dredging – turbidity; tourism – disturbance of animals by noise and production of pollutants and litter; loading of nutrients – eutrophication). All the functions mentioned above can be converted in terms of indices that assess a certain change in ecosystem quality and a certain contribution to the functioning and welfare of human society.

In addition to the above, the application possibilities of other network tools, in use in the social sciences, should also be investigated. Which of the available techniques is most helpful in analysing what part of the social network is responsible for what (eco)system stress as well as the revenues in terms of the production of 'goods and services' and income. This then represents an important part of the economic system.

[insert Fig. 9 here]

3.7.2 Weighting the outcome

Environmental change processes are multifaceted and therefore decision support systems (DSS) need to be comprehensive enough to accommodate a range of decision criteria reflecting the various worldviews and cultural norms that may be present in any given policy context. Economic tools (CBA/CEA) informed by the best available science will need to be buttressed by other tools whenever the policy context is 'contested', which is most often the case in coastal and marine management. We have highlighted earlier the need to incorporate and further develop multi-criteria decision tools and stakeholder engagement processes. A range of techniques have been developed encompassing both risk ranking and more generalised multi-criteria decision making procedures which have evolved around scaling and weighting protocols (with experts and/or stakeholders) (Clemen, 1996; Morgan et al., 2000). Biodiversity assets and ecosystem functioning are hard to value (see for example OECD, 2002), as people recognize their importance and intrinsic value, but are often not able to put a number to it (e.g. Ehrenfeld, 1988). Due to the inherent complexity of valuing these ecosystem attributes and wetlands integral functioning, the data and results obtained from researches, such as those conducted by ENA studies, could be enclosed into the economic valuation process, like for example, during the survey elaboration and hypothetical markets (both in 'willingness to pay' and 'willingness to accept' scenarios) construction, allied to surrounding activities and economic drivers inclusion. By doing so, the study could exemplify the integration of human economic activities and social awareness in general, their relation with biodiversity and ecosystem functioning aspects, and the total social / ecological / economic) value that the system under consideration represents (Fig. 10).

[insert Fig. 10 here]

The information gathered from the ENA analysis is essential to analyse disruptions in coastal systems. However, this information must then be combined with the economic and social information for the successful management of coastal ecosystems. Data integration from the several spheres (ecological, economic and social) is of crucial importance in the design of a useful environmental information system (Fedra, 1997). This integration can then be achieved through the use of tools designed for decision support systems approaches. In this sense, and after the scoping and data gathering stages, multi-criteria analysis (MCA) tools can be used to assist decisions processes. This kind of tool (mostly based on mathematical algorithms) may be flexible and wide-spectrum enough to mitigate the multi-faceted decision problems often associated with ecosystem management and development pressures (see for example Figueira et al., 2005).

Once this (E)NA based approach has been implemented, the scientific and social dialogue around this subject should be maintained. Despite the integrative efforts, some questions remain still open: Are the species chosen to the ENA approach really reflecting the system dynamics at the habitat level? What are suitable reference conditions? What are the boundaries of the system to develop accurate management scenarios? Which measures should be recommended to achieve those scenarios? Are these measures driven by conservational or services-based perspectives?

However, we are the opinion that the proposed approach can facilitate the debate among managers, society and scientific community, creating a common ground for further discussions and developments. Moreover, it should be recalled that, although the habitat scale has been chosen (which allows for a more comprehensive integration of data) it is still 'localised', and although may represent a significant portion of a system, most of the times an ecosystem is composed by several habitats, creating a range of fluxes and interactions that have to be taken into consideration (Fig. 9).

4. Planning the future

The current lack of information and /or progress on better functional and conceptual relationships within the integral system does not mean that no progress can be made in terms of improving decision making and management. The incorporation of system-related indicators and the ecosystem services concept can clarify the relationships between ecosystem change and valued outcomes in terms of 'user' benefits and can therefore lead to better decision making in terms of identifying options with the best net returns to society.

Attention needs to be focused on developing better modelling of ecosystem changes that can be linked to real world management actions (i.e. specific management-related questions) rather than only as the outcome of natural system dynamics. Analysis and indicators need to be targeted on things that managers can influence and recognise as within their competence to change.

Expressing modelling outcomes, through interdisciplinary collaboration, in terms of final ecosystem services and valued human benefits makes the management choice decisions 'easier' by providing a common and readily understood unit of account. Even if not all benefits can be expressed in monetary terms a partial analysis can still improve current decision making e.g. quite often it is possible to demonstrate that the benefits (monetary) of only some of the services provided by an ecosystem(s) in a particular context outweigh the costs of conservation and management, or some development alternative that requires ecosystem removal or severe degradation.

Given the data and conceptual limitations we currently face, it is important to, in many cases, avoid the 'do nothing response' and intervene with the best available 'science' but within an adaptive management strategy that seeks as far as is feasible to keep options open and avoid irreversible change, in a ‘learning by doing’ process.

From the above it is evident that the basis for this sort of approaches is a ‘tailor made’ monitoring program. In another paper (de Jonge et al., 2006) an analysis was made on the development of the water quality monitoring in the United Kingdom and The Netherlands. In that paper it has been proposed to change part of the current monitoring program from a ‘station oriented’ one into an ‘area oriented’ one. As explained there extensively this is something that can be done in a cost effective way and also without ruining the current data series that in The Netherlands already exist since the early 1970s.

5. Conclusions

The diagram in Fig. 10, originally produced by de Groot et al. (2002), has proven to be instrumental for our conclusions in the present context.

From an ecological point of view we have 3 recommendations:

1. Further application of ecological network analysis applied to an ecosystem (species or functional groups) which results in characterizing the system functioning (input-output analysis, system throughput and cycling of compounds) and the magnitude of ENA indices under more or less natural conditions is imperative.
2. Application of the DPSIR approach to the ecosystem (species or functional groups) to quantify in both a dynamical simulation model as well as a network approach (static or dynamic network) the same characteristics and indices as under point 1 can facilitate progress.
3. Applying DPSIR to an aggregated network at the level of the HABIMAP habitat, where the functioning is now compartmentalized according to the habitat criteria seems to be a useful way forward.

From a socio- economic point of view we have 6 recommendations.

4. Clearly define the scale of action: are we valuing global, regional or local assets, and over what time period? To guarantee an accurate description and identification of the relevant services as part of the integral system care must be taken to set the analysis in an appropriate spatial context(including socio-economic, political/governance and cultural conditions).
5. When (e)valuating the provision of the services identified in the network analysis the double counting problem needs special attention.

6. Depending on the type of service, several types of monetary valuation measures may be used. Suitable techniques may be the production function approach, the contingent valuation method (CVM)/ choice experiments etc. But it should also be noted that it is still not possible to meaningfully capture monetary values for all ecosystem services.

7. While we can use the outcomes from economic valuation methods as inputs into a cost-benefit analysis (CBA) in many real world management situations involving difficult trade-off decisions, an overall Multi Criteria Analysis (MCA) will be required if ‘trust’ and ‘accountability’ concerns have to be countered.

8. Coupling of different hierarchical levels and social components by network analysis approaches may result in a multidimensional picture of a social network interacting with some of the economic components impacting the environment and *vice versa*. This picture can be extended by the interaction between the social components and the ecological system. This will need to be better informed by findings now emerging from the behavioural sciences which have highlighted the complexity of human motivations and behaviour.

9. By understanding the role of ecosystem functioning and ecosystem services provision to human wellbeing it is possible to identify and target the natural assets of a system and so accomplish for sustainable development requirements.

To facilitate further progress.

10. When applying ecological network analysis at the habitat level, a ‘tailor made’ monitoring program covering ecological and socio-economic aspects in sufficient detail is essential to guarantee the desired and required progress in valuing, weighing and deciding upon how to proceed in practising sustainable development.

ACKNOWLEDGEMENTS

The present work has benefited from a grant by FCT (SFRH/BD/39093/2007) to Rute Pinto.

6. REFERENCES

Adami, C., 2002. What is complexity? BioEssays 24, 1085-1094.

Anonymous, 1985. Biological research Ems-Dollard estuary. Rijkswaterstaat Communications no. 40/1985, The Hague.

Atkins, J.P., Burdon, D., Elliott, M., Gregory, A.J., 2011. Management of the Marine Environment: Integrating Ecosystem Services and Societal Benefits with the DPSIR Framework in a Systems Approach. Mar. Pollut. Bull. 62, 215-226.

Baird, D., Ulanowicz, R.E., 1993. Comparative study on the trophic structure, cycling and ecosystem properties of four tidal estuaries. Mar. Ecol.-Prog. Ser. 99, 221-237.

Baird, D., McGlade, J.M., Ulanowicz, R.E., 1991. The comparative ecology of six marine ecosystems. Philos. T. R. Soc. B. 333, 15-29.

Baird, D., Asmus, H., Asmus, R., 2004. Energy flow of a boreal intertidal ecosystem, the Sylt-Rømø Bight. Mar. Ecol.-Prog. Ser. 279, 45-61.

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R.E., Jenkins, M., Jefferiss, P., Jessamy, V., Madden, J., Munro, K., Myers, N., Naeem, S., Paavola, J., Rayment, M., Rosendo, S., Roughgarden, J., Trumper, K., Turner, R.K., 2002. Economic reasons for conserving wild nature. Science 297, 950-953.

Balmford, A., Fisher, B., Green, R., Naidoo, R., Strassburg, B., Turner, R.K., Rodrigues, A., 2011. Bringing Ecosystem Services into the Real World: An Operational framework for assessing the Economic Consequences of losing Wild Nature. Environ. Resour. Econ. 48, 161-175.

Barbier, E.B., Koch, E.W., Silliman, B.R., Hacker, S.D., Wolanski, E., Primavera, J., Granek, E.F., Polasky, S., Aswani, S., Cramer, L.A., Stoms, D.M., Kennedy, C.J., Bael, D., Kappel, C.V., Perillo, G.M.E., Reed, D.J., 2008. Coastal Ecosystem-Based management with Nonlinear Ecological Functions and Values. Science 319, 321-323.

Barbier, E.B., Baumgärtner, S., Chopra, K., Costello, C., Duraiappah, A., Hassan, R., Kinzig, A.P., Lehmann, M., Pascual, U., Polasky, S., Perrings, C., 2009. The valuation of ecosystem services, in: Naeem, S., Bunker, D.E., Hector, A., Loreau, M., Perrings, C. (Eds.), Biodiversity, Ecosystem functioning, & Human wellbeing: an ecological and economic perspective. Oxford University Press Inc., New York, pp. 248-262.

Baretta, J.W., Ruardij, P. (Eds.), 1988. Tidal flat estuaries: Simulation and Analysis of the Ems Estuary. Ecological Studies 71, Springer Verlag, Berlin.

Bateman, I.J., Day, B., Georgiou, S., Lake, I., 2006. The aggregation of environmental benefit values: welfare measures, distance decay and total WTP. Ecol. Econ. 60, 450-460.

Bateman, I.J., Mace, G., Fessi, C., Atkinson, G., Turner, R.K., 2011. Economic Analysis for Ecosystem Service assessments. Environ. Resour. Econ. 48, 177-218.

Belfiore, S., Balgos, M., McLean, B., Galofre, J., Blaydes, M., Tesch, D., 2003. A Reference Guide on the Use of Indicators for Integrated Coastal Management. UNESCO, Paris.

Berkes, F., Folke, C., 1998. Linking social and ecological systems. Management practices and social mechanisms for building resilience. Cambridge: Cambridge University Press.

Berkes, F., Colding, J., Folke, C., 2003. Navigating Social-Ecological Systems: building resilience for complexity and change. Cambridge University Press.

von Bertalanffy, L., 1969. General System theory: Foundations, Development, Applications, New York: George Braziller.

Beukema, J.J., 1985. Zoobenthos survival during severe winters on high and low tidal flats in the Dutch Wadden Sea, in: Gray JS, Christiansen ME (Eds.) Marine biology of polar regions and effects of stress on marine organisms. John Wiley, Chichester, pp. 351–361.

Blanchet, H., Lavesque, N., Ruellet, T., Dauvin, J.C., Sauriau, P.G., Desroy, N., Desclaux, C., Leconte, M., Bachelet, G., Janson, A.-L., Bessineton, C., Duhamel, S., Jourde, J., Mayot, S., Simon, S., de Montaudouin, X., 2008. Use of biotic indices in semi-enclosed coastal ecosystems and transitional waters habitats - Implications for the implementation of the European Water Framework Directive. Ecol. Indic. 8, 360-372.

Boyd, J., Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. Ecol. Econ. 63, 616-626.

Borgatti, S.P., 2002. NetDraw: Graph Visualization Software. Analytic Technologies, Harvard.

Borja, A., Franco, J., Pérez, V., 2000. A marine biotic index to establish the ecological quality of soft-bottom benthos within European estuarine and coastal environments. Mar. Pollut. Bull. 40, 1100-1114.

Bossel, H., 1992. Real structure process description as the basis of understanding ecosystems. Ecol. Modell. 63, 261-276.

Bossel, H., 2000. Policy assessment and simulation of actor orientation for sustainable development. Ecol. Econ. 34, 337-355.

ten Brink, B.J.E., Hosper, S.H., Colijn, F., 1991. A quantitative method for description and assessment of ecosystems: The AMOEBA-approach. Mar. Pollut. Bull. 23, 265-270.

Brock, W.A., Finnoff, D., Kinzig, A.P., Pascual, U., Perrings, C., Tschirhart, J., Xepapadeas, A., 2009. Modelling biodiversity and ecosystem services in coupled ecological-economic systems, in: Naeem, S., Bunker, D.E., Hector, A., Loreau, M., Perrings, C. (Eds.), Biodiversity, Ecosystem functioning, & Human wellbeing: an ecological and economic perspective. Oxford University Press Inc., New York, pp. 263-278.

Brundtland, G.H., 1997. The Scientific Underpinning of Policy. Science 277, 457.

Chainho, P., Chaves, M.L., Costa, J.L., Costa, M.J., Dauer, D.M., 2008. Use of multimetric indices to classify estuaries with different hydromorphological characteristics and different levels of human pressure. Mar. Pollut. Bull. 56, 1128-1137.

Clemen, R. T., 1996. Making hard decisions: An introduction to decision analysis, second ed., Duxbury Press, Belmont, CA.

Connell, J.H., 1978. Diversity in tropical rain forests and coral reefs. Science 199, 1302–1310.

Costanza, R., 2000. Social goals and the valuation of ecosystem services. Ecosystems 3, 4-10.

Costanza, R., Low, B.S., Ostrom, E., Wilson, J., 2001. Institutions, Ecosystems, and Sustainability. CRC Press. 270p.

Costanza, R., Daly, H.E., 1992. Natural Capital and Sustainable Development. Conserv. Biol., 6, 37-46.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253-260.

Custance, J., Hillier, H., 1998. Statistical issues in developing indicators of sustainable development. J. R. Statist. Soc. A 161, 281-290.

Dauvin, J.C., Ruellet, T., 2007. Polychaete/amphipod ratio revisited. Mar. Pollut. Bull. 55, 215-224.

Dale, V.H., Beyeler, S.C., 2001. Challenges in the development and use of ecological indicators. Ecol. Indic. 1, 3-10.

Dalsgaard, J.P.T., Oficial, R.T., 1995. Insights into the ecological performance of agroecosystems with ECOPATH II. NAGA. The ICLARM Quart. 18, 26-27.

Daly, H.E., Farley, J., 2004. Ecological Economics: Principles and applications. Island Press, Washington, D.C.

Dame, R.F., Prins, T.C., 1997. Bivalve carrying capacity in coastal ecosystems. Aquat. Ecol. 31, 409-421.

DeAngelis, D. L. 1980. Energy flow, nutrient cycling, and ecosystem resilience. Ecology 61, 764-771.

DETR, 2000. Multi-criteria analysisa - a manual., DETR Appraisal Guidance, Department of the Environment, Transport and the Regions, Eland House, London.

Duarte, C.M., 2000. Marine biodiversity and ecosystem services: an elusive link. J. Exp. Mar. Biol. Ecol. 250, 117-131.

Duarte, P., Meneses, R., Hawkins, A.J.S., Zhu, M., Fang, J., Grant, J., 2003. Mathematical modelling to assess the carrying capacity for multi-species culture within coastal waters. Ecol. Model. 168, 109-143.
Duit, A., Galaz, V., 2008 Governing Complexity – Insights and Emerging Challenges. Governance 21, 311-335.
EC, 2000. Directive 2000/60/EC of European Parliament and of the Council of 23 October 2000 establishing a framework for community action in the field of water policy. Off. J. Eur. Comm., L327, 22/12/200:0001-0073.
EC, 2002. Towards a strategy to protect and conserve the marine environment. Communication from the Commission to the Council and the European Parliament, European Commission, Brussels, 02.10.2002, COM(2002) 539 final.
EC, 2005a. Thematic strategy on the protection and conservation of the marine environment. Communication from the Commission to the Council and European Parliament, European Commission, Brussels, 24.10.2005, COM. (2005) 504 final.
EC, 2005b. Framework for Community Action in the field of marine environmental policy (Marine Strategy Directive). Proposal for a Directive from the European Commission, Brussels, 24.10.2005, COM. (2005) 505 final.
EC, 2008. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). Off. J. Eur. Comm., L 164/19-40.
Ehrenfeld, D., 1988. Why put a value on biodiversity, in: Wilson, E.O. (Ed.), Biodiversity. National Academy Press, Washington, DC.
Ehrlich, P.R., Mooney, H.A., 1983. Extinction, substitution, and ecosystem services. Bioscience 33, 248-254.
Elliott, M., Quintino, V., 2007. The Estuarine Quality Paradox, Environmental Homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. Mar. Pollut. Bull. 54, 640-645.
Fath, B.D., 2004. Network analysis in perspective: comments on ‘‘WAND: an ecological network analysis user-friendly tool’’. Environ. Modell. Softw. 19, 341-343.
Fedra, K., 1998. Integrated Risk Assessment and Management: Overview and State-of-the-Art. Journal of Hazardous Materials 61, 5-22.
Figueira, J., Greco, S., Ehrgott, M. (Eds.), 2005. Multiple Criteria Decision Analysis: State of the Art Surveys. Springer Science+Business Media, Inc., New York.

Finn, J.T., 1976. Measures of ecosystem structure and function derived from analysis of flows. J. Theor. Biol. 56, 363–380.

Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. Ecol. Econ. 68, 643-653.

Goodland, R.J.A., 1987. The World Bank's wildlands policy: A major new means of financing conservation. Conserv. Biol. 1, 210-213.

Griffin, J.N., O'Gorman, E.J., Emmerson, M.C., Jenkins, S.R., Klein, A.M., Loreau, M., Symstad, A., 2009. Biodiversity and the stability of ecosystem functioning, in: Naeem, S., Bunker, D.E., Hector, A., Loreau, M., Perrings, C. (Eds.), Biodiversity, Ecosystem functioning, & Human wellbeing: an ecological and economic perspective. Oxford University Press Inc., New York, pp. 78-93.

de Groot, R.S., Wilson M.A., Boumans R.M.J., 2002. A typology for the classification, description, and valuation of ecosystem functions, goods and services. Ecol. Econ. 41, 393-408.

de Groot, R.S., Alkemade, R., Braat, L., Hein, L., Willemen, L., 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol. Compl. 7, 260-272.

Grover, J.P., 1997. Resource Competition. Chapman and Hall, London.

Gunderson, L.H., Holling, C.S., 2002. Panarchy: Understanding transformations in systems of humans and nature. Island Press, Washington, DC.

Haag, D., Kaupenjohann, M., 2001. Parameters, prediction, post-normal science and the precautionary principle - a roadmap for modelling for decision-making. Ecol. Model. 144, 45–60.

Hannon, B., 1973. The structure of ecosystems. J. Theor. Biol. 41, 535-546.

Hardi, P., Zdan, T., 1997. Assessing Sustainable Development: Principles in Practice. IISD.

Heal, G., Kristrom, B., 2005. National Income and the Environment, in: Mäler, K.G., Vincent, J. (Eds.), Handbook of Environmental Economics, Elsevier, Amsterdam, Vol. 3., Chp 22.

Hellawell, J.M., 1986. Biological indicators of fresh water pollution and environmental management. Elsevier Applied Science Publishers, New York.

Herman, P.M.J., Middelburg, J.J., Widdows, J., Lucas, C.H., Heip, C.H.R., 2000. Stable isotopes as trophic tracers: combining field sampling and manipulative labelling of food resources for macrobenthos. Mar. Ecol.-Prog. Ser. 204: 79–92.

Holling, C.S., 1978. Adaptive environmental assessment and management. John Wiley & Sons, New York.

Holling, C.S., Meffe, G.K., 1996. Command and control and the pathology of natural resources management. Conserv. Biol. 10, 328-337.

Horn, H.S., 1975. Markovian properties of forest succession, in: Cody, M.L., Diamond, J.M. (Eds.), Ecology and Evolution of Communities. Belknap Press, Cambridge, Massachusetts, pp. 196–211.

Huisman, J., Weissing, F.J., 1999. Biodiversity of plankton by species oscillations and chaos. Nature 402, 407–410.

Huisman, J., Weissing, F.J., 2001. Fundamental unpredictability of multispecies competition. Am. Nat. 157, 488–494.

Janssen, R., 1994. Multiobjective Decision Support for Environmental management. Kluwer, Dordrecht.

Janssen, M.A., Bodin, Ö., Anderies, J.M., Elmqvist, T., Ernstson, H., McAllister, R.R.J., Olsson, P., Ryan, P., 2006. Toward a network perspective of the study of resilience in social-ecological systems. Ecol. Soc. 11, article 15. URL:http://www.ecologyandsociety.org/vol11/iss1/art15/

de Jong, D.J., 2000a. Definition of terms in habitat world. Unpublished Paper, RIKZ, The Hague.

de Jong, D.J., 2000b. Ecotopes in the Dutch Marine Tidal Waters: A proposal for a classification of ecotopes and a method to map them. C.M. - International Council for the Exploration of the Sea, CM 2000 (T:05). ICES [S.l.].

de Jonge, V.N., 2007. Toward the application of ecological concepts in EU coastal water management. Mar. Pollut. Bull. 55, 407-414.

de Jonge, V.N., Boynton, W., D'Elia, C.F., Elmgren, R., Welsh, B.L., 1995. Comparison of effects of eutrophication on North Atlantic estuarine systems, in: Dyer, K.R. and D'Elia, C.F. (Eds.), Changes in fluxes in estuaries, pp. 179-196.

de Jonge, V.N., Kolkman, M.J., Ruijgrok, E.C.M., de Vries, M.B., 2003. The need for new paradigms in integrated socio-economic and ecological coastal policy making. Proceedings of 10th International Wadden Sea Symposium, Ministry of Agriculture, Nature Management and Fisheries, Department North, Groningen, pp. 247-270.

de Jonge, V.N., Elliott, M., Brauer, V.S., 2006. Marine monitoring: its shortcomings and mismatch with the EU Water Framework Directive's objectives. Mar. Pollut. Bull., 53, 5–19.

Johnson, J.C., Luczkovich, J.J., Borgatti, S.P., Snijders, T.A.B., 2009. Using social network analysis tools in ecology: Markov process transition models applied to the seasonal trophic network dynamics of the Chesapeake Bay. Ecol. Model. 220, 3133-3140.

Jørgensen, S.E., 1997. Integration of ecosystem theories: a pattern. Second ed., Kluwer Academic Publishers, Dordrecht.

Jørgensen, S.E., 2010. Ecosystem services, sustainability and thermodynamic indicators. Ecol. Compl. 7, 311-313.

Jörgensen, S.E., Müller, F., 2000. Handbook of Ecosystem Theories and Management. CRC Press, New York.

Knoflacher, M., Gigler, U., Tötzer, T., Naefe, B., 2003. Assessment of Sustainability – Can it be standardised? Proceedings of the EASY-ECO-2 Evaluation of Sustainability – European Conferences, Vienna.

Kremen, C., Merenlender, A.M., Murphy, D.D., 1994. Ecological monitoring: a vital need for integrated conservation and development programs in the tropics. Conserv. Biol. 8, 388–397.

Kristensen, N.P., Gabric, A., Braddock, R., Cropp, R., 2003. Is maximizing resilience compatible with established ecological goal functions? Ecol. Modell. 169, 61-71.

Krugman, P., 2009. How did economists get it so wrong, New York Times, 20th Sept.

Layard, R., 2010. Measuring subjective well being. Science 327, 534-535.

Lenton, T. M., 1998. Gaia and natural selection. Nature 394, 439-447.

Leontief, W., 1951. The Structure of the American Economy, 1919–1939. second ed., Oxford University Press, New York.

Levin, S., 1999. Fragile domination: complexity and the commons. Perseus Book, Reading, MA.

Levrel, H., Fontaine, B., Henry, P.-Y., Jiguet, F., Julliard, R., Kerbiriou, C., Couvet, D., 2010. Balancing state and volunteer investment in biodiversity monitoring for the implementation of CBD indicators: a French example. Ecol. Econ. 69, 1580-1586.

Likens, G., 1992. An ecosystem approach: its use and abuse. Excellence in ecology. Book 3. Ecology Institute, Oldendorf/Luhe, Germany.

Limburg, K.E., O'Neill, R.V., Costanza, R., Farber, S., 2002. Complex systems and valuation. Ecol. Econ. 41, 409-420.

Lindeman, R.L., 1942. The trophic dynamic aspect of ecology. Ecology 23, 399-418.

Luczkovich, J.J., Borgatti, S. P., Johnson, J. C., Everett, M. G., 2003. Defining and measuring trophic role similarity in food webs using regular equivalence. J. Theor. Biol. 220, 303-321. see also http://drjoe.biology.ecu

Ludwig, D., B. Walker, and C. S. Holling. 1997. Sustainability, stability, and resilience. Conservation Ecology,1, art. 7. URL: http://www.consecol.org/vol1/iss1/art7/

Luisetti, T., Turner, R.K., Bateman, I.J., Morse-Jones, S., Adams, C., Fonseca, L., 2011. Coastal and marine ecosystem services valuation for policy and management: managed realignment case studies in England. Ocean Coast. Manage. 54, 212-224.

McMahon, S.M., Miller, K.H., Drake, J., 2001. Networking tips for Ecologists and Social Scientists. Science 293, 1604-1605.

Marques, J.C., Basset, A., Brey, T., Elliot, M., 2009. The ecological sustainability trigon-a proposed conceptual framework for creating and testing management scenarios. Mar. Pollut. Bull. 58, 1773-1779.

Martínez-López, B., Perez, A.M., Sánchez-Vizcaíno, J.M., 2009. Social network analysis. Review of general concepts and use in preventive veterinary medicine. Transbound. Emerg. Dis. 56, 109-120.

McLusky, D.S., Elliott, M., 2004. The estuarine ecosystem: ecology, threats and management, third ed., Oxford University Press, Oxford.

Millennium Ecosystem Assessment (MEA), 2003. Ecosystems and Human Well-Being: a framework for assessment. Island Press, Washington, DC.

Mitchell, R.C., Carson, R.T., 1989. Using surveys to value public goods: The contingent valuation method. Resources for the Future, Washington, DC.

Morgan, M.G., Florig, H.K., Dekay, M.L., Fischbeck, P., 2000. Categorizing risks for risk taking. Risk Analysis 20, 49-58.

Naeem, S., Bunker, D.E., Hector, A., Loreau, M., Perrings, C., 2009. Introduction: the ecological and social implications of changing biodiversity. An overview of a decade of biodiversity and ecosystem functioning research, in: Naeem, S., Bunker, D.E., Hector, A., Loreau, M., Perrings, C. (Eds.), Biodiversity, Ecosystem functioning, & Human wellbeing: an ecological and economic perspective. Oxford University Press Inc., New York, pp. 3-13.

Norgaard, R.B., Kallis, G., Kiparsky, M., 2009. Collectively engaging complex socio-ecological systems: re-envisioning science, governance, and the California Delta. Environ. Sci. Policy 12, 644–652.

Nunes, P.A.L.D., van den Bergh, J.C.J.M., 2001. Economic valuation of biodiversity: sense or nonsense? Ecol. Econ. 39, 203-222.

Odum, E.P., 1969. The strategy of ecosystem development. Science 164, 262-270.

Odum, E.P., 1971. Fundamentals of ecology, third ed., Saunders, Philadelphia, PA.

OECD, 1994. Environmental Indicators: OECD Core Set, Paris: Organisation for Economic Co-operation and Development.

OECD, 2002. Handbook of Biodiversity Valuation. A guide for policy makers. OECD, Paris.

van Oevelen, D., Soetaert, K., Middelburg, J.J., Herman, P.M.J., Moodley, L., Hamels, I., Moens, T., Heip, C.H.R., 2006. Carbon flows through a benthic food web: Integrating biomass, isotope and tracer data. J. Mar. Res. 64, 453-482.

Olson, D.L., 1996. Decision aids for selected problems. Springer, New York.

Ormerod, P., 2010. N squared-public policy and the power of networks. RSA, London. (www.RSA.org/about-us/RSA-pamphlets).

Patrício, J., Ulanowicz, R., Pardal, M.A., Marques, J.C., 2004. Ascendency as an ecological indicator: a case study of estuarine pulse eutrophication. Estuar. Coast. Shelf S. 60, 23-35.

Patrício, J., Neto, J.M., Teixeira, H., Salas, F., Marques, J.C., 2009. The robustness of ecological indicators to detect long-term changes in the macrobenthos of estuarine systems. Mar. Environ. Res. 68, 25-36.

Pearson, T.H., Rosenberg, R., 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanogr. Mar. Biol. Annual Review 16, 229-311.

Perrings, C., Baumgä rtner, S., Brock, W.A., Chopra, K., Conte, M., Costello, C., Duraiappah, A., Kinzig, A.P., Pascual, U., Polasky, S., Tschirhart, J., Xepapadeas, A., 2009. The economics of biodiversity and ecosystem services, in: Naeem, S., Bunker, D.E., Hector, A., Loreau, M., Perrings, C. (Eds.), Biodiversity, Ecosystem functioning, & Human wellbeing: an ecological and economic perspective. Oxford University Press Inc. New York, pp. 230-247.

Pinto, R., Patrício, J., Baeta, A., Fath, B.D., Neto, J.M., Marques, J.C., 2009. Review and evaluation of estuarine biotic indices to assess benthic condition. Ecol. Indic. 9, 1-25.

Pinto, R., Patrício, J., Neto, J.M., Salas, F., Marques, J.C., 2010. Assessing estuarine quality under the ecosystem services scope: Ecological and socioeconomic aspects. Ecol. Compl. 7, 389-402.

Rogers, S.I., Greenaway, B., 2005. A UK perspective on the development of marine ecosystem indicators. Mar. Pollut. Bull. 50, 9-19.

Rosenberg, R., Blomquist, M., Nilsson, H.C., Cederwall, H., Dimming, A., 2004. Marine quality assessment by use of benthic species-abundance distribution: a proposed new protocol within the European Union Water framework Directive. Mar. Pollut. Bull. 49, 728-739.

Salas, F., 2002. Valoración y aplicabilidad de los índices e indicadores biológicos de contaminación orgánica en la gestión del medio marino. Tesis de doctorado. Murcia: Universidad de Murcia.

Salas, F., Marcos, C., Neto, J.M., Patrício, J., Pérez-Ruzafa, A., Marques, J.C., 2006. User-friendly guide for using benthic ecological indicators in coastal and marine quality assessment. Ocean Coast. Manage. 49, 308-331.

Simboura, N., Zenetos, A., 2002. Benthic indicators to use in ecological quality classification of Mediterranean soft bottom marine ecosystems, including a new biotic index. Mediterr. Mar. Sci. 3, 77-111.

Singh, R.K., Murty, H.R., Gupta, S.K., Dikshit, A.K., 2009. An overview of sustainability assessment methodologies. Ecol. Indic. 9, 189-212.

Smaal, A.C., Prins, T C., Dankers, N., Ball, B., 1998. Minimum requirements for modeling bivalve carrying capacity. Aquat. Ecol. 31, 423-428.

Sutinen, J.G., 2007. Socioeconomics and the Ecosystem Approach to Management of Marine Resources. Report to NOAA, University of Rhode Island, Kingston.

Thébault, J., Schraga, T.S., Cloern, J.E., Dunlavey, E.G., 2008. Primary Production and Carrying Capacity of Former Salt Ponds After Reconnection to San Francisco Bay. Wetlands 28, 841-851.

Thrush, S.F., Hewitt, J.E., Norkko, A., Nicholls, P.E., Funnell, G.A., Ellis, J.I., 2003. Habitat change in estuaries: predicting broad-scale responses of intertidal macrofauna to sediment mud content. Mar. Ecol.-Prog. Ser. 263, 101-112.

Tilman, D., 1982. Resource Competition and Community Structure. Princeton University Press, Princeton.

Turner, R.K., 1993. Sustainability, Resource Conservation and Pollution Control: An Overview, in: R. Kerry Turner (Ed.), Sustainable Environmental Economics and Management: Principles and Practice. London, Belhaven Press, pp: 1−36.

Turner, R.K., 2000. Integrating natural and social science in coastal management. J. Marine Syst. 25, 447-460.

Turner, R.K., Lorenzoni, I., Beaumont, N., Bateman, I.J., Langford, I.H., McDonald, A.L., 1998. Coastal management for sustainable development: analysing environmental and socio-economic changes on the UK coast. The Geographical Journal 164, 269-281.

Turner, R.K., Paavola, J., Farber, S., Jessamy, V., Georgiou, S., 2003. Valuing Nature: Lessons Learned and Future Research Directions. Ecol. Econ. 64, 292-510.

Turner. R.K., Hadley, D., Luisetti, T., Lam, V., Cheung, W., 2010. An introduction to Socio-Economic Assessment within a Marine Strategy Framework. DEFRA, London.

Ulanowicz, R.E., 1980. An hypothesis on the development of natural communities. J. Theor. Biol. 85, 223-245.

Ulanowicz, R.E., 1986. A phenomenological perspective of ecological development. In Poston, T.M., Purdy, R. (Eds.), Aquatic Toxicology and Environmental Fate. Vol. 9, ASTM STP 921. American Society for Testing and Materials, Philadelphia, pp. 73-81.

Ulanowicz, R.E., 1997. Ecology, the Ascendent Perspective. Columbia University Press, New York.

Ulanowicz, R.E., 1999. NETWRK 4.2a: A package of computer algorithms to analyse ecological flow networks. Solomons, MD.

Ulanowicz, R.E., and Wulff. 1991. Comparing ecosystem structures: the Chesapeake Bay and the Baltic Sea, in: J. Cole, G. Lovett, and S. Findlay (Eds.), Comparative analysis of ecosystems, pattern, mechanism, and theories. Springer-Verlag, New York.

Vitousek, P.M., Mooney, H.A., Lubchenco, J., Melillo, J.M., 1997. Human domination of Earth's ecosystems. Science 277, 494-499.

Wasserman, S., Faust, K., 1994. Social Network Analysis. Cambridge University Press.

Ysebaert, T., Meire, P., Herman, P.M.J., Verbeek, H., 2002. Macrobenthic species response surfaces along estuarine gradients: prediction by logistic regression. Mar. Ecol.-Prog. Ser. 225, 79-95.

Ysebaert, T., Herman, P., 2002. Scale-dependent predictive modelling of estuarine soft-sediment benthic macrofauna. NIOO-KNAW Report 2002-6, Yerseke, The Netherlands.

Zonta, R., Guerzoni, S., de Jonge, V.N., Pérez-Ruzafa, A., 2007. Measuring and managing changes in estuaries and lagoons: morphological and eco-toxicological aspects. Mar. Pollut. Bull. 55, 403-406.

LEGENDS TO FIGURES

Fig. 1.

Diagram representing the general structure of the ‘integral system’ without explicit incorporation of the human/social component (reproduced from Zonta et al., 2007). Influences of natural factors are indicated by dashed lines and those by anthropogenic activities by solid lines. A_1 and A_2 refer to the natural or anthropogenic influence of the physical system, B_1 and B_2 to that of the physico-chemical system, and C refers to direct human effects on the biological system.

Fig. 2.

Diagram as in Fig. 1 but now extended with over time changing system ‘expressions’ and visualised variations in abundance at the species level.

Fig. 3.

Illustration of a step by step construction of a habitat map based on the factors salinity, current velocity, depth and emergence period. The procedure is applied to a real estuary, the Westerschelde (southwestern part of The Netherlands) resulting in 14 different habitats covering the marine and the brackish part of the estuary.

Fig. 4.

Reproduction of the food web of the Sylt-Rømø Bight as published by Baird et al. (2004). The food web represents 59 compartments aggregated into an 18 compartments model used for ecological network analysis. Numbers in bold face: aggregation numbers; numbers in brackets: original compartment numbers; thin arrows: interactions between compartments; thick arrows: pelagic interactions. For detailed information the reader is referred to the above cited publication.

Fig. 5.

Diagram showing the relationship between some of the ecosystem indices as used in ecological network analysis (ENA).

Fig. 6.

A strongly simplified food web illustrating the main fluxes of energy (often proxied by e.g. carbon, phosphorus or nitrogen) between the different trophic levels.

Fig. 7.

Diagram as presented in Fig. 1 but now the physical, physico-chemical and biological subsystems are substituted by estuarine zones or compartments and connected habitats. Further, the generalisations from the figures 1 and 2 have been substituted by realistic examples (channel maintenance dredging, loading of the system by pollutants and nutrients and fisheries. Moreover, the DPSIR framework has been applied where D= driver, P = pressure to system, S = state of system, I = impact to humans and R = the supposed human response. The parts of the system which represent the 'intermediate services' and the 'final services' or 'human benefits' are also indicated.

Fig. 8.

Description of potential habitat related properties and designated human activities to indicate the suitability of the habitat level as the possible ecological and economic integration level.

Fig. 9.

Visualisation of the combination of the information in the Figs 7 and 8 where (ecological) network analysis is carried out per habitat and then coupled to relevant pressures from the socio-economic system. The total sum of the state changes and related impact to the socio-economic system will result in an integrated human response to the total impact.

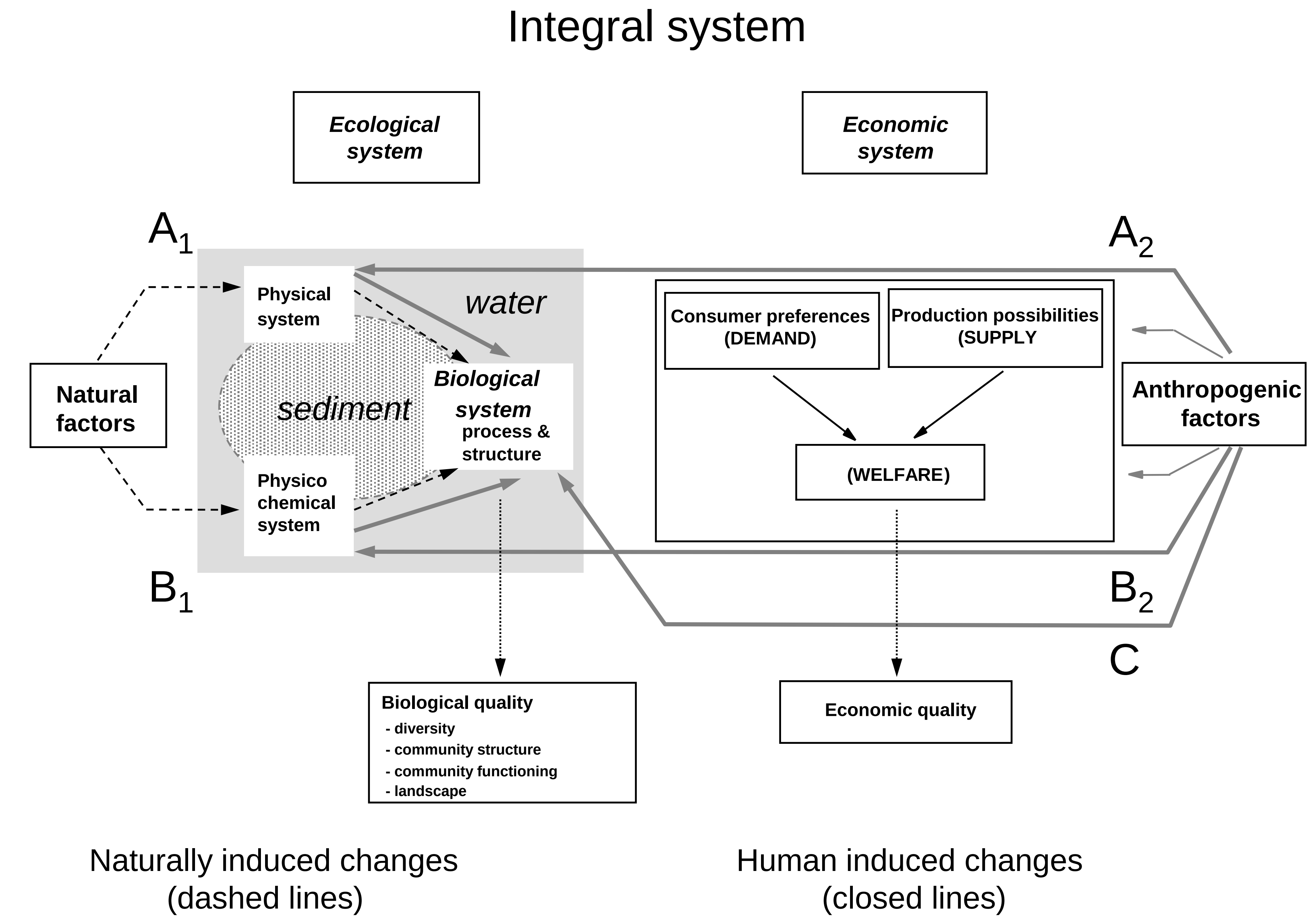
Fig. 10.

Overview of the components representing the functioning, the functions and the value of parts of the integral system as well as the total system. The total system value is composed of the sum of the ecological, social-cultural and economic values. (modified after de Groot et al., 2002)

***4 Ethical Statement**

Ethical Guidelines for Journal Publication - We accept these as displayed on the OCMA site and further emphasize that all three authors have explicitly contributed to this piece of work.

Figure 1



Integral system
Ecological system
Economic system
A1
A2
water
Physical system
sediment
Biological system
process & structure
Physico chemical system
Natural factors
Consumer preferences (DEMAND)
Production possibilities (SUPPLY)
(WELFARE)
Anthropogenic factors
B1
B2
C
Biological quality
- diversity
- community structure
- community functioning
- landscape
Economic quality
Naturally induced changes (dashed lines)
Human induced changes (closed lines)

Figure 2

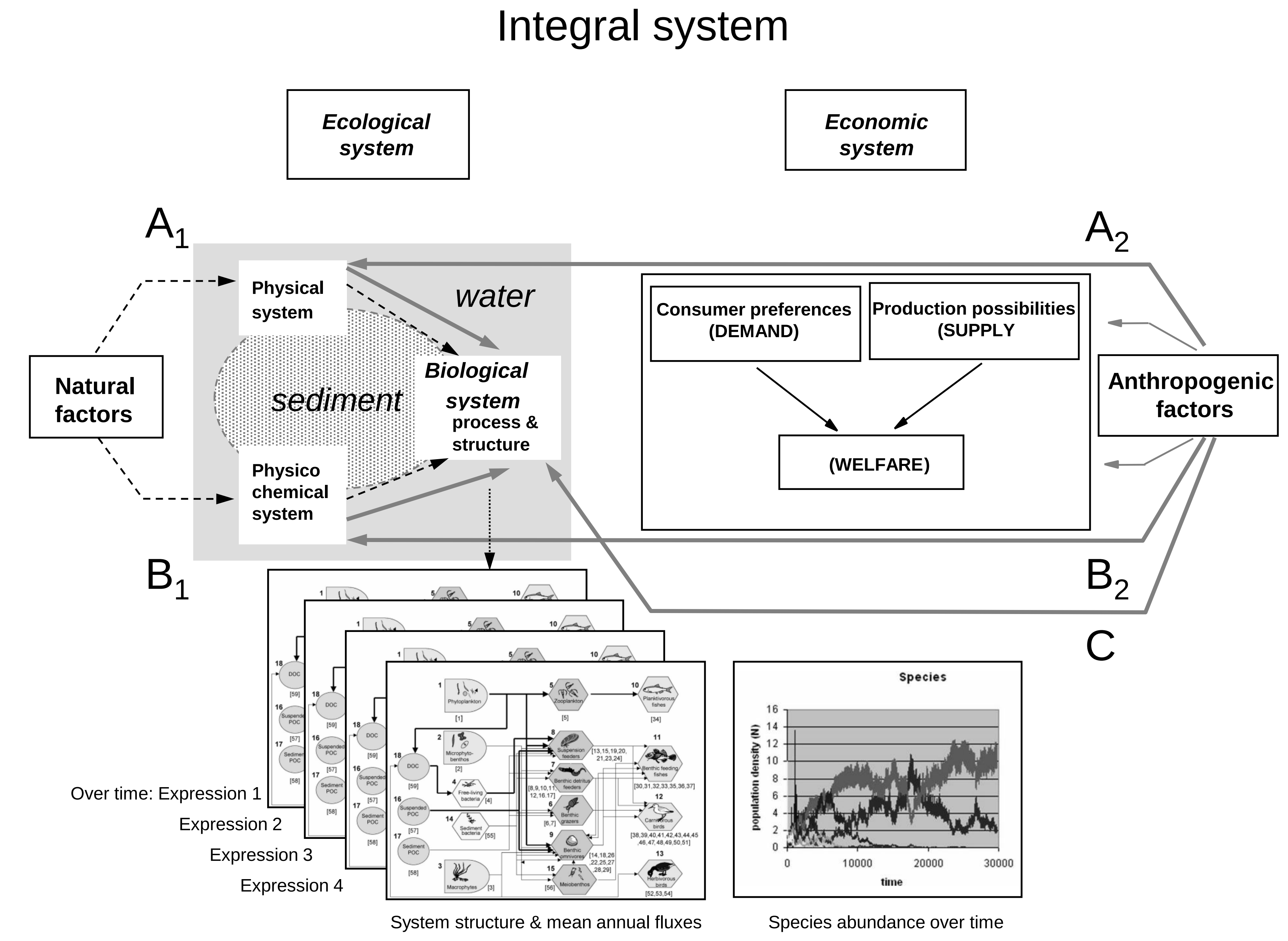
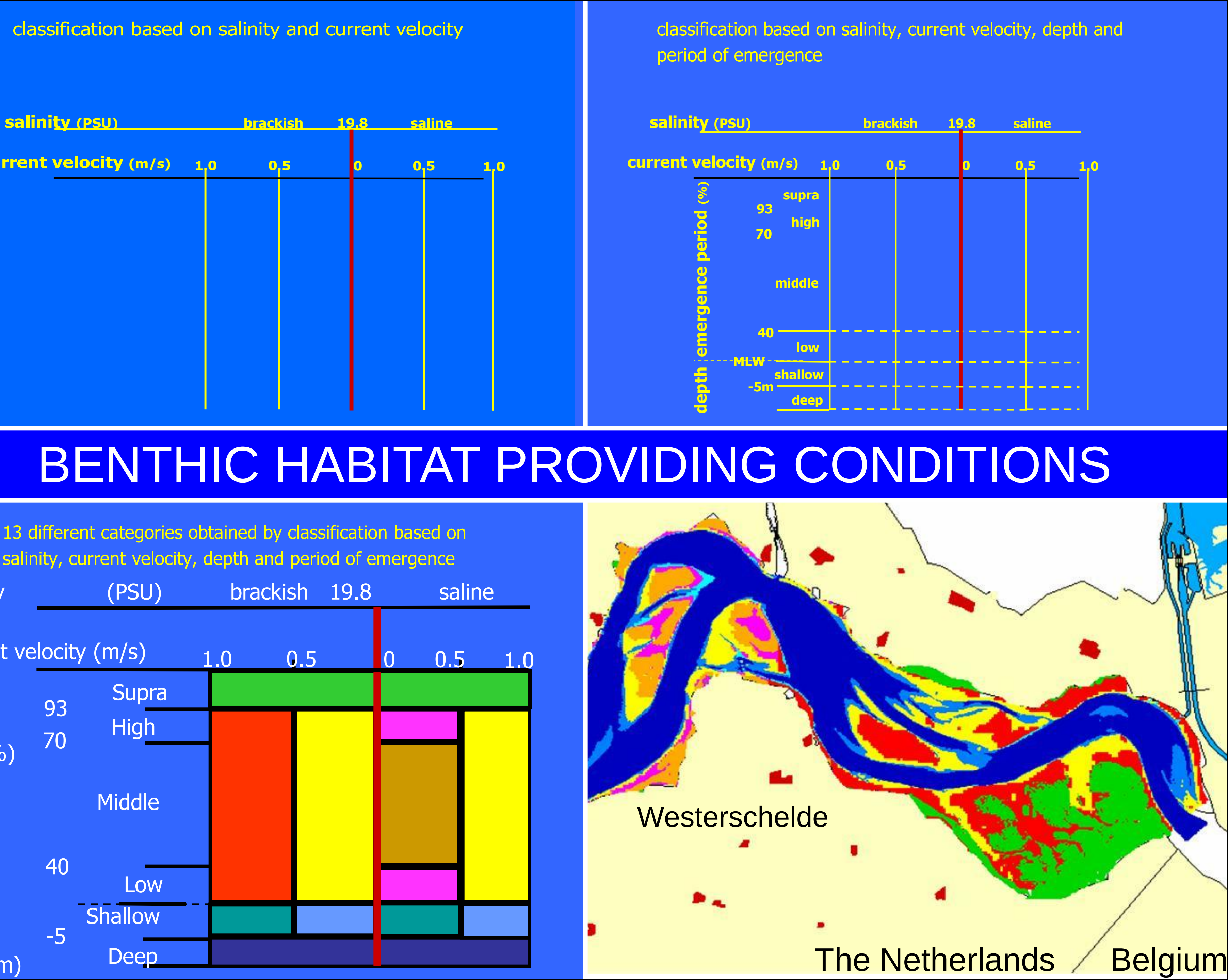

Integral system
Ecological system
Economic system
A1
A2
B1
B2
C
Natural factors
Physical system
water
sediment
Physico chemical system
Biological system process & structure
Consumer preferences (DEMAND)
Production possibilities (SUPPLY)
(WELFARE)
Anthropogenic factors
Over time: Expression 1
Expression 2
Expression 3
Expression 4
Species
population density (N)
time
System structure & mean annual fluxes
Species abundance over time

Figure 3



classification based on salinity and current velocity
salinity (PSU)
brackish
19.8
saline
current velocity (m/s)
1.0
0.5
0
0.5
1.0
classification based on salinity, current velocity, depth and period of emergence
salinity (PSU)
brackish
19.8
saline
current velocity (m/s)
1.0
0.5
0
0.5
1.0
depth emergence period (%)
supra
93
high
70
middle
40
low
MLW
shallow
-5m
deep
BENTHIC HABITAT PROVIDING CONDITIONS
13 different categories obtained by classification based on salinity, current velocity, depth and period of emergence
salinity
(PSU)
brackish
19.8
saline
current velocity (m/s)
1.0
0.5
0
0.5
1.0
Supra
93
High
70
(%)
Middle
40
Low
MLW
Shallow
-5
Deep
(m)
Westerschelde
The Netherlands
Belgium

Figure 4

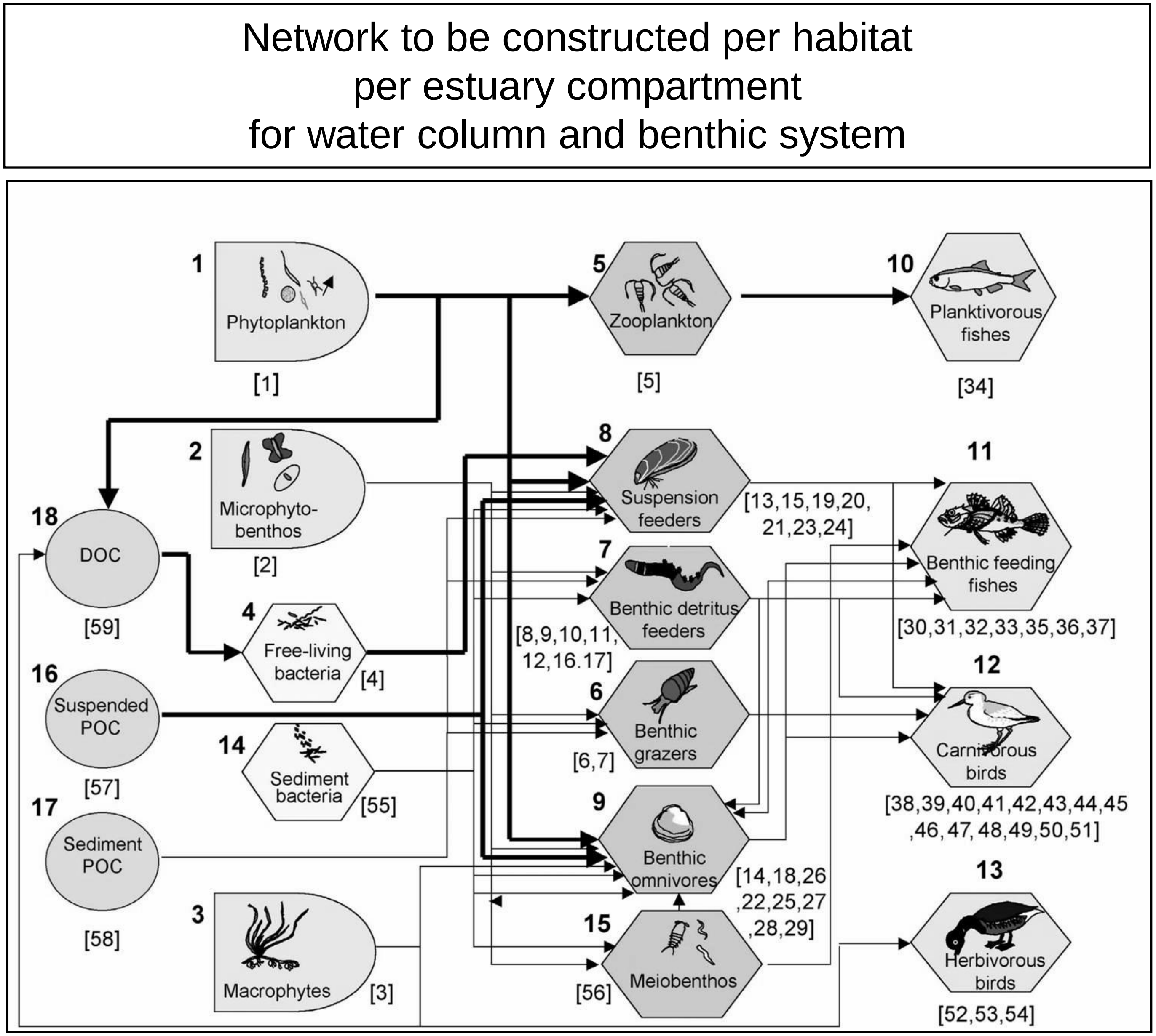
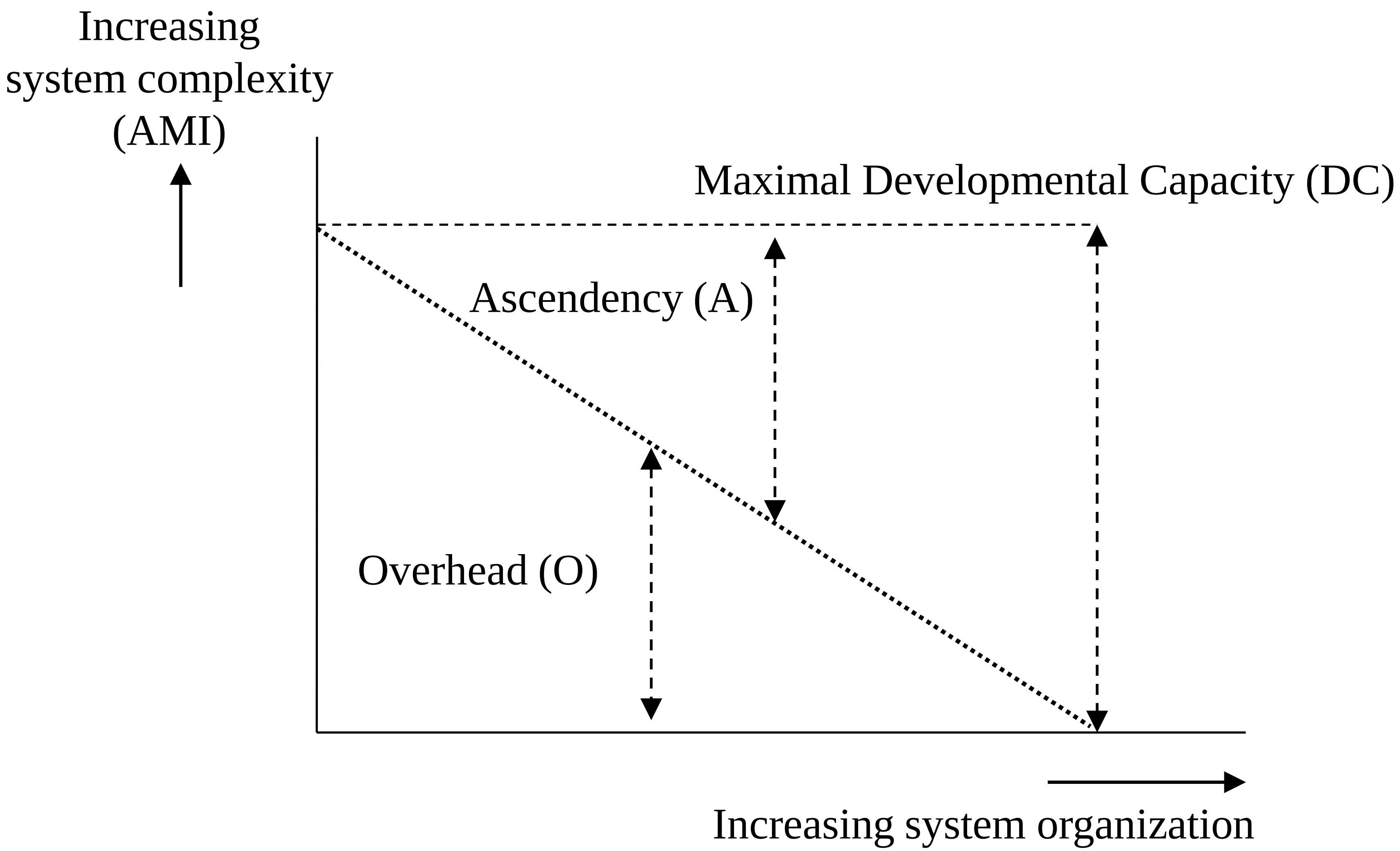

Network to be constructed per habitat
per estuary compartment
for water column and benthic system
1
Phytoplankton
[1]
5
Zooplankton
[5]
10
Planktivorous fishes
[34]
2
Microphyto-benthos
[2]
8
Suspension feeders
[13,15,19,20, 21,23,24]
11
Benthic feeding fishes
[30,31,32,33,35,36,37]
18
DOC
[59]
7
Benthic detritus feeders
[8,9,10,11, 12,16.17]
4
Free-living bacteria
[4]
16
Suspended POC
[57]
6
Benthic grazers
[6,7]
12
Carnivorous birds
[38,39,40,41,42,43,44,45 ,46,47,48,49,50,51]
14
Sediment bacteria
[55]
17
Sediment POC
[58]
9
Benthic omnivores
[14,18,26 ,22,25,27 ,28,29]
3
Macrophytes
[3]
15
Meiobenthos
[56]
13
Herbivorous birds
[52,53,54]

Figure 5



Increasing
system complexity
(AMI)
Maximal Developmental Capacity (DC)
Ascendency (A)
Overhead (O)
Increasing system organization

Figure 6

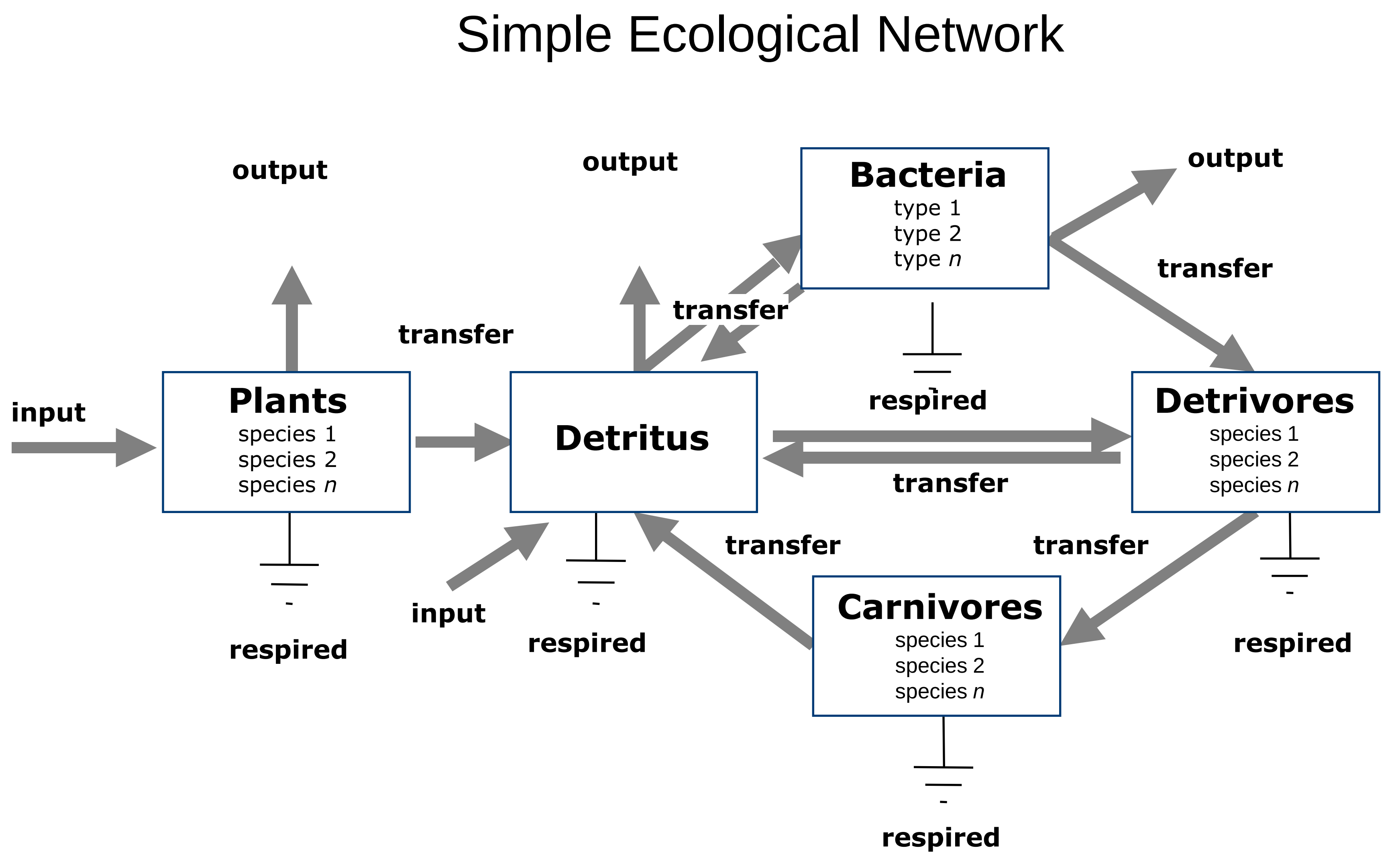
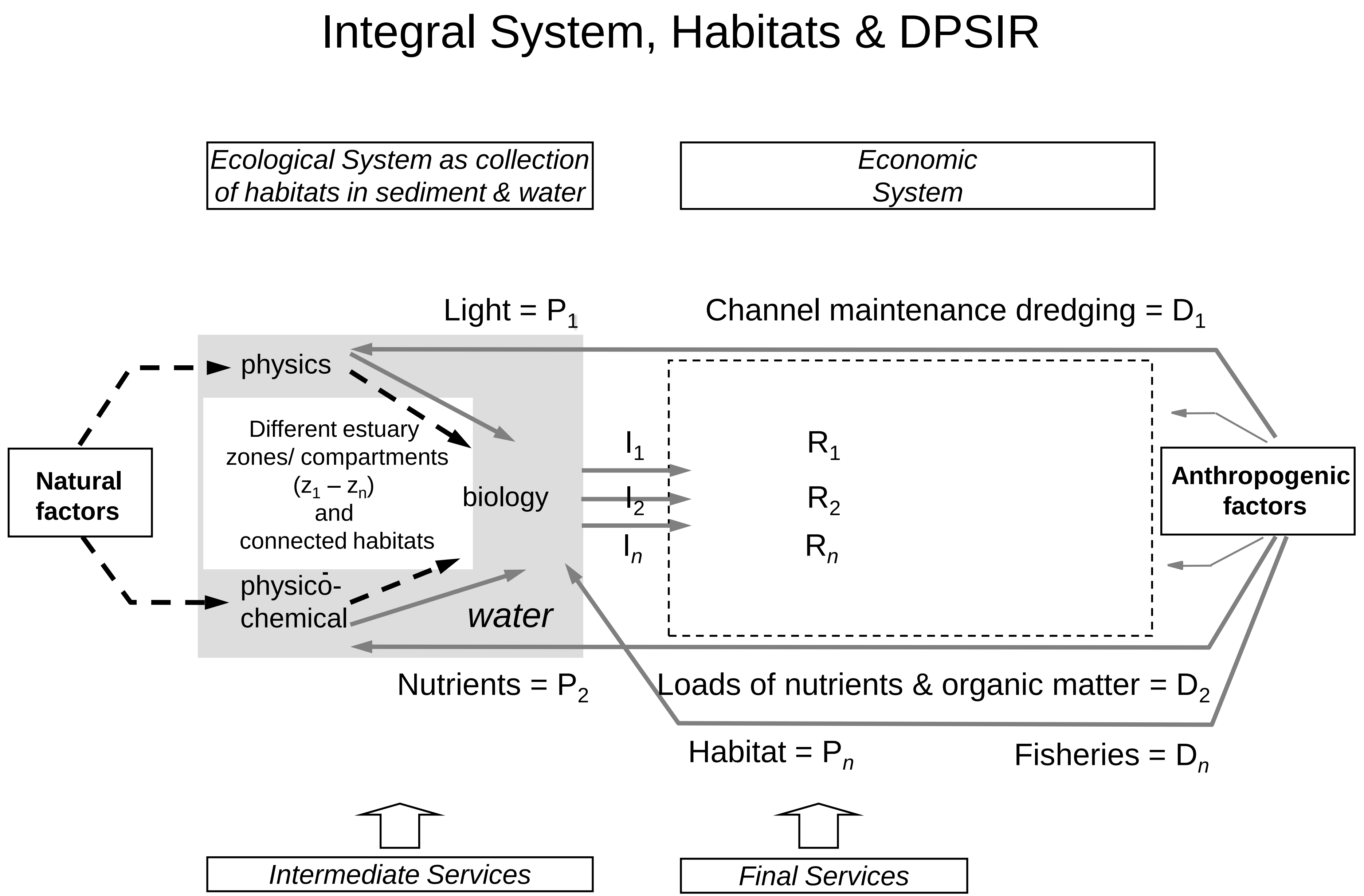

Simple Ecological Network
output
output
Bacteria
type 1
type 2
type n
output
transfer
transfer
transfer
input
Plants
species 1
species 2
species n
Detritus
respired
Detrivores
species 1
species 2
species n
transfer
transfer
transfer
input
respired
respired
Carnivores
species 1
species 2
species n
respired
respired

Figure 7



Integral System, Habitats & DPSIR
Ecological System as collection of habitats in sediment & water
Economic System
Light = P_1
Channel maintenance dredging = D_1
physics
Different estuary zones/ compartments ($z_1 – z_n$) and connected habitats
biology
physico-chemical
water
Natural factors
I_1
I_2
I_n
R_1
R_2
R_n
Anthropogenic factors
Nutrients = P_2
Loads of nutrients & organic matter = D_2
Habitat = P_n
Fisheries = D_n
Intermediate Services
Final Services

Figure 8

Coupling of habitat to human functions

Habitat related properties			Human activities
Elevation level of habitat	Habitat type		
Sublittoral: main channels	Deep channels as part of the water column	←	shipping, dredging
Intertidal: creeks	Low and high turbulent intertidal creeks as part of the water column	←	fishing
Intertidal: high, midddle, low	High, middle and low littoral benthic system	←	culturing of mussel and oyster, shrimp fishery
Supralittoral: saltmarshes	Supralittoral benthic system	←	grazing cattle, sheep & horses, eco-tourism, bird watching

:

Figure 9

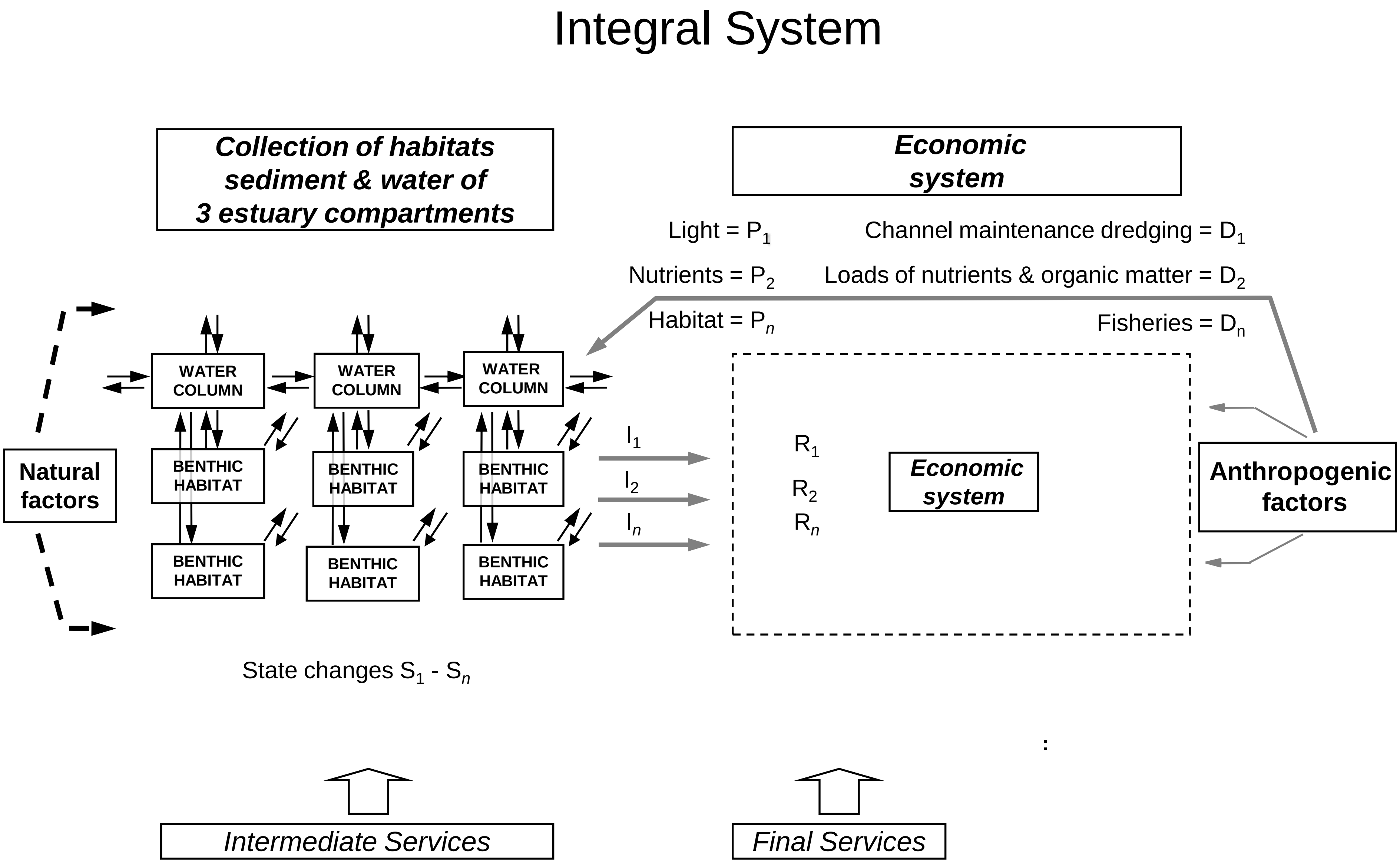

Integral System
Collection of habitats sediment & water of 3 estuary compartments
Economic system
Light = P1
Nutrients = P2
Habitat = Pn
Channel maintenance dredging = D1
Loads of nutrients & organic matter = D2
Fisheries = Dn
WATER COLUMN
WATER COLUMN
WATER COLUMN
BENTHIC HABITAT
BENTHIC HABITAT
BENTHIC HABITAT
BENTHIC HABITAT
BENTHIC HABITAT
BENTHIC HABITAT
Natural factors
I1
I2
In
R1
R2
Rn
Economic system
Anthropogenic factors
State changes S1 - Sn
Intermediate Services
Final Services

Figure 10

Valuing the Integral System