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Hendrik Schubert · Felix Müller *Editors*

Southern Baltic Coastal Systems Analysis

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Foreword

Since two decennia, coastal research in Germany is promoted mainly through funding of the Federal Research Ministry. Since one decennium I am retired so I was honoured to be asked to give my comments on this substantial contribution to coastal research encompassing disciplinary studies on aquatic and marine habitats as well interdisciplinary studies including ecosystem services on such land–water transition areas. Globally coastal habitats are important areas for nature, as well as for humankind. Therefore interest in understanding these habitats is growing.

During and after reading through this immense complex matter, I was left with a series of impressions that I will present and discuss in a more or less arbitrary order. Coastal research has been strongly promoted since the early 2000s. ICZM (Integrated Coastal Zone Management) was the magic word. Most of the initial studies were disciplinary oriented (geology, marine and aquatic biology, etc.). Later on and impressively demonstrated in the BACOSA and SECOS projects interests changed from disciplinary to multi- and inter-disciplinary studies. In other words from the disciplinary natural science approaches to the human oriented ecosystem services assessment approaches, a major and risky activity.

Coastal research in Germany can be considered as fragmented, mainly due to the different institutions that are involved. Their main interest is natural science of the environment, e.g. the Baltic Sea, the North Sea and the Wadden Sea or the oceanic waters including Arctic and Antarctic waters. The main interference with the human dimension takes place in the coastal zone where human activities interact, mostly in a negative way (pollution, space claims) with the natural environment. This is well illustrated in several chapters of the book. Looking at a broader perspective, one would like to see a much more intensive cooperation between all the institutes involved through guiding activities of the KDM (Consortium of German Marine Research), where almost all institutes are represented. However, as far as I know, the KDM has not been involved in prioritization of research proposals or stimulated future research directions. Because of the large amount of expertise available, the establishment of a National German Institute for Coastal Research would have been a challenging idea. A department of coastal terrestrial studies should be included to cover all aspects of ecosystem services and human interactions.

The most challenging issue in this book is the attempt to construct a method for the assessment of ecosystem services. A long and intensive text is needed to explain all the different aspects of ecosystem services, which comprise supporting, regulating, provisioning and cultural services. All four parts are well documented and discussed, new methods to assess them described.

What is missing is a financial evaluation method (see, e.g., Costanza et al. 1997, de Groot et al. 2010). An exception is the attempt to use willingness to pay in a context of touristic use of the coast. This could have been a useful extension towards policy makers and coastal managers. A future activity taken into account should be the application of the ecosystem services assessment in a concrete case with an environmental problem to be solved. I strongly suggest to come up with possibilities to simplify the type of presentation of results of such assessments to make them more understandable for a group of non-experts. This brings me to a déjà vu from my occupation in the Netherlands. In preparation of a new water strategy plan, we were asked to come up with a method to assess ecosystem health. We developed a method based on the occurrence of about 30 species from low to high in the food web. Their numbers or densities were compared between nowadays and a reference period (about 1930, if data available). The current data were expressed against the reference values in a radar plot which delivered an 'amoeba' type of diagram, very easy to understand to which extent numbers differed from the reference. The Minister of Infrastructure and Public Works herself presented the diagram at a symposium on the North Sea (see ten Brink et al. 1991).

In one of the last chapters, the role of the EU in setting coastal and marine policies is well illustrated through the complexity of coming up with new alternatives during a relatively short period. These policy issues are rather confusing.

Finally, the authors have done a brilliant job by starting with an overarching set of questions at the very beginning and answering them in a convincing way at the end. This book will be a milestone in discussion about the human interaction in the coastal zone.

Büsum, Germany
January 2022

Franciscus Colijn

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Preface

The specific smell of a harbour market, a mixture of tar, paint and fish, is regarded as an extremely attractive holiday element, but beach wrack is mostly seen as a nuisance—e.g. because of its smell. In- and out-coming cruise ships can gather large enthusiastic crowds of spectators, whereas others, mainly residents, are complaining about the exhausted pollution. Moreover, even within rather homogeneous groups the points of view about a specific aspect or a specific coastal structure may differ largely. For instance, high biodiversity is seen as a positive feature in general, but when it comes to the establishment of neozooans and neophytes, even in unsaturated ecosystems as the Baltic, other perceptions are arising quickly.

Faced with these different opinions, local and regional authorities must find balanced solutions in their multiple decision-making processes. They are exposed to a large, sometimes contrasting spectrum of judgements, expectations and interests of stakeholders, all of them highlighting their very specific target aspects and armed with their very specific arguments. During the respective debates, soon emotions come into play, stirred up by lobbyist groups and hampering constructive discussions. So where is the neutral ground for environmentally sustainable and at the same time socially accepted solutions? Searching for a solution and relying on the (still) high reputation of science as a source of objective knowledge, decision makers are asking for sound and intersubjective arguments to withstand the pressures from lobbyists.

Consequently, after several decades of intensive research along the German Baltic coast, funded by a broad spectrum of regional, national as well as international agencies, reliable and robust knowledge about all aspects of coastal ecosystems from economic aspects via cultural and societal approaches to natural sciences' results should be ready at hand. However, as seen during, e.g., the establishment of assessment schemes for the EU-Water Framework Directive or, later, for the Marine Strategy Directive, our knowledge is still fragmented. Discipline-related approaches have revealed many new details and have given rise to a bunch of new concepts and challenging hypotheses—but large gaps have been left open between the knowledge-related home ranges of the different disciplines. For example, we knew a lot about tourist behaviour at coastal holiday resorts, but almost nothing was known about their prevalence with respect to the beach appearance. Details of nutrient cycling and relations to ecosystem structure, irrespective of some extant uncertainties, have been studied a lot at the terrestrial as well as the aquatic part of coastal systems—but the interactions between these two subsystems were at the very best regarded as input/output parameters, ignoring the enormous variabilities or the important feedback mechanisms.

In general, the lack of exchange between the disciplines resulted in a fragmented knowledge base where data provided by one discipline did not fulfil the requirements of the other; so no comprehensive concept for describing coastal ecosystem functionality in a holistic manner was available. In addition, to overcome this situation, during the past decades a series of truly interdisciplinary research projects tried not only to gather the existing knowledge and to fill in disciplinary gaps, but also to develop a concept for assessing the effects of various simultaneous anthropogenic impacts on the system's states.

For the German Baltic coast, this work was done, e.g., by the sister projects BACOSA (Baltic Coastal System Analysis and Status Evaluation) and SECOS (Understanding and Quantifying the Scope and Scale of Sedimentary Services in the German Baltic Sea) as parts of the KÜNO research program (Küstenforschung Nord- und Ostsee), aiming to analyse the interplay between anthropogenic pressures, ecosystem status and climatic factors. Both projects did not start at scratch, disciplinary parts of them just filled in remaining substantial knowledge gaps rather than developing brand-new concepts of ecosystem function. However, closing the gaps, thought to be proof of existing concepts, resulted in some surprising new insights, e.g. about limitation patterns of eutrophic coastal water bodies. But the main aim, and consequently also the red line of this book, was to develop an instrument bridging the gap between scientific results, gathered by disciplinary analysis, and societal demand for a comprehensive knowledge base for well-balanced management decisions. Consequently, writers and readers are facing the challenge to combine a brought arc of knowledge, from deep philosophy to, e.g., exact hydrochemistry.

In filling up this interdisciplinary bow, many colleagues and friends have been helpful in conceiving and realizing this book. Therefore, we wish to thank all of these persons for support, assistance and encouragement. Especially we wish to thank

- The colleagues and supporters from both sequences of the projects BACOSA and SECOS.
- The colleagues and coordinators of the KÜNO research programs between 2013 and 2019.
- The funding institutions from BMBF and PTJ, as well as the home-universities and institutions of the authors.
- The authors of the following papers for their constructive cooperation, enthusiasm, patience, optimism and skilfulness.
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Rostock, Germany — Hendrik Schubert
Kiel, Germany — Felix Müller
January 2022

Contents

Part I

Introduction

1 Southern Baltic Coastal Systems Analysis: Questions, Conceptions, and Red Threads

Hendrik Schubert and Felix Müller

Abstract

This chapter sets the frame for the book by introducing the reader into the motivation for coastal ecosystem research at the Southern Baltic Sea, a region under increasing pressure caused by anthropogenic impact. Especially the last decades transformed them to an extent, that serious concerns about their functionality raised requests for sustainable management schemes. This chapter outlines the societal demands behind these developments and the research programs conducted to solve the problems along the path to societally accepted management decisions.

The overall increasing *anthropogenic impacts* in the environment have not only been altering coastal systems, but have also resulted in a steadily increasing number of conflicts of interests. In this situation, the governmental bodies are asked for balanced decisions, respecting the individual interests of various stakeholders and interest groups. Doing this, the weighting of multiple arguments requires solid science-based reasons. And the decision makers need *interdisciplinary approaches* in order to respect the economic, cultural, ecological, and social aspects, which are intertwined in sustainable management strategies. Such comprehensive interdisciplinary studies have been conducted in the past for several terrestrial systems, but they are rather scarce with respect to *marine and coastal systems*.

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The respective modern terrestrial examples have demonstrated that *ecosystem services (ESS)* can be suitable instruments for integrating science with the social and economic aspects of sustainability for a balanced recognition of the various related aspects and requirements. ESS have therefore shown a promising potential for providing a platform for constructive discussions. However, in order to do so, ESS–assessments must be performed on a sound and broad knowledge base. To realize this requirement, all involved disciplines should provide the instrument with data reflecting a deep understanding of the *systems structures and functions* before analyzing interactions and feedback loops together. The resulting *forecast-potential* is the main focus which such instrument is asked for, e.g., to serve as a platform to assess anticipated changes. Climate change impacts, coastal protection measures, installations of windfarms, aquaculture, eutrophication, technical installations, or local development measures are examples for the rising management demands in coastal environments. The desirable local forecast scenarios can be developed by comparison with already existing sites; however, the knowledge base *of the historical status* in most of the cases is not complete, but restricted to specific aspects only. This especially applies to ecological data in the marine realm, where thorough analysis of ecosystem structure and function was not done before industrialization whereas data about economic circumstances, cultural as well as social aspects are at least existing. But "existing" does neither mean that they are easily available nor that they have been analyzed in a suitable approach. A comprehensive history of southern Baltic coastal regions, dealing with all aspects of cultural and economic developments during the past centuries is still missing irrespective of the large number of region- or town-specific publications. Summarizing, for *economic and cultural aspects data exist*, which needed to be analyzed whereas for *ecological aspects data deficiency* has to be substituted by in-depth analysis of the functionality of the recent system. This is what the two projects *BACOSA and SECOS* were aiming to contribute to, in order to improve the knowledge base for the development of an instrument for spatial planning, respecting all aspects of a sustainable use of coastal ecosystems.

Within that situation, this book documents some interesting parts of the outcomes of the German research program *KÜNO*, which has been following the target to "improve the scientific basis for ecosystem-oriented, sustainable management of coastal resources and to make its results available to practice-oriented users,"[1] referring to the German coastal ecosystems of the North Sea and the Baltic Sea. As components of that program, the two Baltic Sea projects *BACOSA* and *SECOS* are providing the main contents of this book. *SECOS* ('The service of sediments and the coastal sea in the German Baltic') has studied the "distribution and quantitative relevance of sedimentary services in the range of the German Baltic waters by measuring, mapping and modelling of future scenarios with the aim to advance the development of management tools." SECOS II[2] has aimed for "a better

[1] https://deutsche-kuestenforschung.de/

[2] https://www.io-warnemuende.de/project/141/secos_ii.html

understanding of transport, exchange and interaction processes between water and sediments, "..." providing mapping tools for areal quantification of structural and biogeochemical properties linked to sediment functions." To do so, SECOS II has aimed for the extension and application of a marine ecosystem-service-evaluation-framework, that covers the German Baltic Sea, spatially integrates natural scientific data, model simulation results and socio-economic aspects into an evaluation tool that visualizes the societal benefits and serves as an umbrella for the integration of marine policies.

The *BACOSA*-Project ('Baltic Coastal System Analysis and Status Evaluation')[3] has aimed "at analysing the quality and quantity of the functions of aquatic plants and has intended to identify and evaluate ecosystem services provided by coastal ecosystems of the Baltic Sea." The aim of *BACOSA II* was to characterize, quantify and value the historical development of ecosystem service supply in the German Baltic Coast region in order to determine the interrelations of ecosystem services with environmental, social, economic and ethical conditions."

The *target of this book* is to integrate important results of these projects and cooperating activities with a special emphasis on interdisciplinarity and linkages between human and environmental coastal sub-systems. We have structured the subsequent steps of knowledge description in this book in 31 chapters, which are each following certain research questions. These questions will be guiding the following introduction and you will find them again in the conclusions of this book. There are of course also *focal questions* concerning the whole contents of this book. These are as follows:

- *Q1: What can we learn from actual case studies of coastal ecosystem analysis in order to evaluate the actual condition of the ecosystems along the German Baltic Sea coastline?*
- *Q2: Is it possible to integrate the multiple aspects of social, ethical and environmental sciences in order to characterize, indicate and measure ecosystem service potentials and flows?*
- *Q3: Is such analyse a useful base for ecosystem management decisions and is it sufficiently significant, robust and applicable to serve as an instrument for sustainability policy?*

To find answers for these queries, we are attempting a *stepwise integration*, which cannot reach up to a total holistic overall view but to the proposal of an interesting pathway how the very different and diverging parts can be brought together. One branch of argumentation will be based on the cooperation of scientific disciplines; another one will be based on the environmental demands for integration and a third pathway will be shown through human-environmental systems approaches. An outcome of this level of integration will be demonstrated by indicator studies on ecosystem services. The basic structure of this conception can be seen in Fig. 1.1.

[3] https://www.ecosystem-management.uni-kiel.de/en/research/projects/bacosa

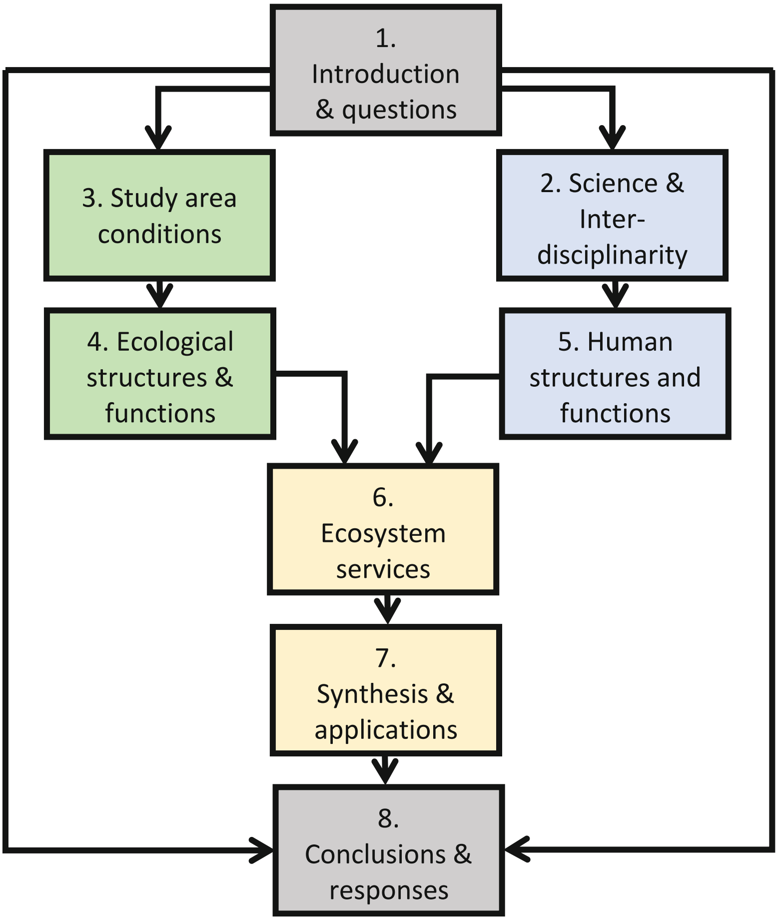

Fig. 1.1 General structure of the book and lines of argumentation

The *second chapter* of this book, nominated as **"Coastal ecosystems from a social-ecological perspective"** (Benkenstein et al., this volume) tries to introduce the different disciplinary viewpoints, which have been denoted before. It is therefore a conceptual text section, which tries to argue for the general integration of individual scientific approaches and starting points. Therefore, Chap. 2 can be understood as a formal and structural description of scientific positions and developments, including an expose of the process of interdisciplinary interaction. The focal guiding question of Chap. 2 is:

- *Q4: What are the demands of coastal research and management for cooperation between the involved scientific disciplines, and how has the attained interdisciplinarity been applied in this book?*

The tentative answers and comments on this question are ordered in the following textual sequence: After an introduction of the single topics, the basic necessity for interdisciplinary approaches in coastal analyses is underlined due to the demand side (Chap. 2) and due to general scientific issues, requirements and reasons for integrative approaches (Chap. 2). These arguments are followed by descriptions of the disciplinary, science-based starting points, their specific targets, demands, methodologies and potential contributions (Chap. 2). Hereby, the authors are discussing aspects from marine and coastal ecology, ecosystem ecology, environmental economics and ethics. Also social and legal aspects are briefly considered, but the focal philosophical approach is described comprehensively from multiple viewpoints, and finally, in Chap. 2 we try to show how this diverse information has been integrated to an interdisciplinary pattern in the framework of this book.

With the *third chapter* (named "**structures and functions of the research area"**), we are moving from theoretical considerations to a practical subject, getting to know the environmental conditions in and around the research area and enfolding the research question 5:

- *Q5: Which are the basic environmental conditions of the research area of the following chapters?*

Thus, here we can find the basic long-term features of the overall study area and the existing habitat types. In the beginning of this brief presentation of the study regions of this book, Chap. 3 (Schubert et al., this volume) demonstrates the abiotic conditions of the southern Baltic Sea. In Chap. 4, Papenmeier and Arz (this volume) provide an introduction of the "geological and sedimentary conditions and their developments." Thereafter Müller et al. (this volume) describe the ecological conditions of the terrestrial hinterland areas of the research region (Chap. 5). Jurasinski et al. (this volume) concentrate on the ecology of the direct coastlines in Chap. 6 and they add a detailed analysis of ecosystem conditions in the reed zones of the Eastern German coast. Finally, the sediment-based habitat structures and the ecological patterns in the marine zones of the Southern Baltic Sea are described by Zettler and Darr (this volume, Chap. 7). This information provides the ground for the more detailed analyses of Chaps. 9–18.

These investigations are described and summarized under a clear ecological focus in the following Chapter, nominated "**Ecological structures and functions of the coastal and offshore water body ecosystems**" by Blindow et al. (this volume). *Chapters 9–18* is targeted on a comprehensive analysis of structure-function links and their variability with respect to limitation patterns in time, ranging from seasonality aspects to decadal long-term trends. For the first time, a synoptical assessment for all sub-systems of a coastal ecosystem allows for direct analysis of interactions and feedback mechanisms. The respective research questions are:

- *Q6: Which are the basic ecosystem mechanisms, interrelations and patterns in the respective habitats and which is their seasonal and long-term variability?*

- *Q7: Can this knowledge help to provide a sound ecological data base for human-environmental systems analysis?*
- *Q8: How do the investigated ecosystems react after human modifications, which is their reactivity, resilience and adaptability?*

Related to these questions, the ecological analysis of the coastal ecosystems begins with an introduction in Chap. 9 (Blindow and Forster, this volume) before Blindow et al. (this volume) start to highlight the special characteristics of coastal lagoons as transient zones between terrestrial and marine influences in Chap. 10. This attitude is reinforced by describing the special study sites of the following paragraphs in Chap. 11 (Schumann and Blindow, this volume). Finally, in Chap. 12. Schumann et al. (this volume) demonstrate significant data about short-term variability, long-term trends and seasonal aspects in the Darß-Zingst Bodden Chain. These ecological items are accomplished by a special study on carbon fluxes and food webs and the effect of macrophytes on the lagoons' food web characteristics by Paar et al. (this volume) in Chap. 13.

In the following four texts, the habitats of the coastal zones of the open Baltic Sea are investigated. The sequence starts with a structural characterization of the benthic habitats by Gogina and Zettler (this volume) in Chap. 14. In the following Chap. 15, the matter fluxes are in focus of the descriptions. Forster et al. (this volume) analyze these functional components concentrating on the important exchange processes by bioturbation. Thereafter, seasonal aspects and short-term variabilities of the offshore ecosystems are discussed by Dutz and Wasmund (this volume) in Chap. 16. The long-term trends of the offshore ecosystems are contained in Chap. 17 (Wasmund and Zettler, this volume) and in the end of the ecophysiological analyses, Berthold (this volume) provides results about nutrient and limitation regimes in coastal water ecosystems in Chap. 18.

With these passages, some significant aspects of the ecological conditions in the coastal waters of the German Baltic Sea are introduced. Therefore, with the subsequent chapters, we are widening the scope and moving into socio-ecological systems. Therefore, in *Chap. 19* (Ott et al., this volume), the human factor is added to the analysis. This accomplishment starts with some theoretical considerations, whereby ethical aspects are playing a major role. Hereby, different approaches to merge the main contrasting lines of human demands by means of ethical and economic points of view are presented and discussed, followed by an introduction into the interdisciplinary approach of human-environmental systems. The respective research question related to "**The human factor—coastal social-ecological systems**" is this:

- *Q9: Which are the focal mechanisms, interrelations and patterns of the societal aspects in order to provide a sound knowledge base for human-environmental systems analysis?*

The resulting depiction of human-environmental systems by Ott et al. (this volume) starts with a short introduction in Chap. 5 and then highlights the systems from an economic aspect in Chap. 5, stressing perspectives from behavioral sciences as a basis for economic environmental activities. These considerations are expanded by a detailed inception of philosophical arguments. The text introduces, discusses, derives and compares the human-environmental relations from different viewpoints such as environmental virtue ethics, eudaimonic theory, biophilia, religion, or inherent moral values in Chap. 5. These concepts are applied in Chap. 5 within the discussion of valuation strategies and methods, which includes a first introduction of the ideas of ecosystem services. Another approach is demonstrated in Chap. 5: here some basic arguments and concepts of general systems analysis and ecosystem analysis are used to discuss the basic outlines of human-environmental systems conceptions, which can serve as a level of integration bridging philosophical, economic, social, and scientific approaches. Finally, the major human-related geographical structures of the research area are mapped in Chap. 5 as a supplement to the ecological descriptions from Chap. 4.

Up to that position, the object of our studies has been described at different levels, from a pure aspect of *theory of science* and the concept of *interdisciplinarity* over an analysis of the spatial, structural and functional *ecological conditions*. It was ending in an application of the interdisciplinary basics from Chap. 2 in Chap. 19, where the *ecological fundament* has been built for a practical integration on the next, applicable level. That is the development of methods in order to indicate the potentials of the southern Baltic ecosystems to provide *ecosystem services*. Thereby, the following questions will be guiding the discussions in the following paragraphs:

- *Q10: Which are the most effective ecosystem services in the research areas, how can they be described and indicated and how can we derive them from ecosystem analysis linked with societal approaches?*

This task is carried out and demonstrated in some case studies of *Chaps. 20–26* of this book (“**Combining the aspects—Ecosystem service assessment”**). It starts with a short introduction (Chap. 20) and a conceptual merging of the arguments discussed before. That is the basis for the review article of Kuhn et al. (this volume, Chap. 21), who are on the search for missing links in ecosystem service research related to the situation of the Baltic Sea. They are showing fields of problems, and one such field becomes obvious in the report of Ott and Berg (this volume) about the cultural services of the lagoons of Mecklenburg-Vorpommern (Chap. 22). Besides some illustrative examples of service provision, the authors express the statement that from an ethical viewpoint, a qualitative description of cultural services is sufficient, while attempts of quantification are connected with problems originating in philosophical attitudes. That this is a rather disciplinary viewpoint is shown by the following papers. Poser/Frank and Benkenstein (this volume) provide a report on economic valuations using conjoint analyses of coastal touristic areas in Chap. 23. A more comprehensive approach is developed by Schumacher et al. in Chap. 24. Here, a terrestrial matrix approach has been supplemented, adapted and applied to assess the ecosystem service potential of marine and terrestrial habitats. This is a new combination, which was applied to different scales of the German Baltic coastline. A compatible approach is demonstrated by Inacio and Schernewski (this volume) in Chap. 25. Here, the authors are concentrating on the temporal dynamics of ecosystem service potentials in different Baltic lagoons by comparing recent and historical data. Also in Chap. 26, we are illuminating ecosystem service dynamics. In this last paper from Bicking et al. (this volume) the viewpoint is directed into the future, describing the potential outcome of some climate and land-use-based scenarios on ecosystem service budgets.

Chapters 27–30 completes the circle of arguments by presenting the feedback from the individual disciplines, demonstrating potential applications and identifying open questions and defining the limits of applicability of ESS. The title is "**Synthesis—valuation as a tool for managing coastal ecosystems"** and the respective questions are related to the overall outcome of this study and potential applications of the derived results and methods. The introduction of this first step of a synopsis (Chap. 27) sets the pace for three articles that try to provide elements of a synthesis of the preceding elaborations. In this sence, Blindow et al. (this volume) are applying the ecosystem service matrix technique to demonstrate the consequences of different ecosystem states in Chap. 28. Thereby it becomes clear how significant the role of macrophytes, phytoplankton and bioturbation can be for the overall capacities to provide ecosystem services. In Chap. 29, Schernewski and Robbe (this volume) are discussing the strategic and instrumental potential of the ecosystem service approach to find a suitable degree of application in coastal management and policy. Finally, Ott et al. (this volume, Chap. 30) discuss the role of ecosystem services for nature protection purposes and for the coastal sustainability concept in general.

In the end, *Chap. 31 ("Conclusions")* provides a brief summary, highlighting the progress achieved and describing fields for application as well as giving recommendations for future research in order to increase the robustness and reliability of forecast potentials within ESS assessments. A focus of the chapter is set on some tentative answers of the research questions, which have been listed beforehand.

Altogether, this volume is thought to offer a comprehensive insight into several aspects of coastal systems of the southern Baltic, ranging from the functioning of ecosystems via socio-economical aspects to ethical concepts. Basing on the obtained results of interdisciplinary research, it addresses transdisciplinary problems and can serve as a sound state-of-the-art knowledge base, stimulating further research as well as being used for the development of management scenarios and strategies with a broad societal acceptance.

Coastal Ecosystems from a Social-Ecological Perspective

2

Martin Benkenstein, Konrad Ott, Michael Rauscher, Hendrik Schubert, and Felix Müller

Abstract

This Chapter has the function of introducing the different starting positions of the authors and to provide a first list of viewpoints on social-ecological systems of the southern Baltic region. After a short general introduction, Chap. 2 describes the central role of human needs for the construction of a unified human-environmental model conception. It argues towards the approach of ecosystem services and gives a first impression on the demand for interdisciplinary and transdisciplinary integration. This strategy is generally deepened in Chap. 2, while in Chap. 2, the situation in different participating disciplines is described: It is shown from which state marine ecology, coastal ecology, ecosystem analysis, environmental economics, and environmental ethics have proceeded to cooperate on the attempt to better understand the coastal systems from a multidisciplinary

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point of view. Finally, the demand for interdisciplinary integration is illuminated in Chap. 2 with respect to the following contents and structures of this book.

2.1 Introduction

In "*The Sea Around US*," Rachel Carson wrote in 1951 about "man":[1] "He cannot control or change the ocean as, in his brief tenancy of earth, he has subdued and plundered the continents" (1951, p. 20). Retrospectively, this quotation echoes a final illusion of planetary infinity. Even if humans cannot "control" the ocean in its entirety, they have changed both the ocean and its coastal zones deeply.

Meanwhile, the ocean is warming due to climate change, ocean acidification has become an encroaching long-term problem, and the intake of substances implies pollution (heavy metals, plastics) and eutrophication. Oxygen-poor regions increase. Specific ecological systems, as coral reefs, are under threat. Some species of marine mammals (blue whales) and fish (tuna) are at the edge of extinction. Coastlines are transformed into human-dominated infrastructures (cities, harbors, tourist destinations, aquacultures, bridges). Many fish stocks are harvested at limits or are overfished. Shipping dominates global trade, adding to marine pollution and to submarine noise, disturbing marine mammals. Deep sea mining is an option of ongoing extractivism. Activities from naval forces add to human impacts upon the ocean. In addition to these degradation factors, we have to consider environmental burdens from the past, as ammunitions from World War II accumulated on the ocean grounds (as in the Baltic Sea) as well as dumping of hazardous substances, which was common in the past century.[2]

The ocean has now been reached by the forces of the Anthropocene. This is true *a fortiori* for the smaller "seas" which are connected to the ocean but are usually surrounded by civilized coastal zones (as the Baltic Sea, the Mediterranean Sea, the Black Sea, and others). Marine conservation and restoration efforts are to be located at different scales. There is planetary or biospheric scale on which we face only one ocean, but there are also continental, national, regional, and even communal scales on which we face specific seas, coastlines, and brackish waters. Environmental policy making also is situated within a multi-layered system: international regimes (as UNFCCC, CBD, CITES, OSPAR, HELCOM etc.), EU policies (as FFH, WRRL), federal state (national), regional, etc. On a global scale, the ocean has been represented in the Sustainable Development Goals (SDG). SDG 14 (misnamed "Life Below Water") has been interpreted from a theoretical "strong" sustainability perspective (Neumann et al. 2017). As Neumann et al. (2017) and Franke et al. (2020) argue the prominent SDG metaphor of "healthy ocean" should be conceived

[1] In 1951, male inclusion was usual in grammar even for a female author. Today, of course, all sex and gender are referred to within inclusionary speech. This is the case in this article.

[2] The dilemma is here that removing the rusted ammunition from the seabed might result in a sudden release of highly toxic substances into the marine environment.

in terms of a principle that one should promote the fertility/productivity, resilience, and richness/diversity of all land- and seascapes. It also holds with respect to the German Baltic coastlines. Under this normative principle, full attention can be devoted to the scope of ecosystem services. The full rationale is given in Chap. 19.

The ecosystem service approach has been applied to different scales in many case studies. Most environmental actions affect primarily but not exclusively minor scales. If we restore the Baltic Sea, we do not affect the Chinese Sea, the Pacific Northwest, and the Gulf of Mexico, but may provide benefits to the Danish North Sea. Minor scales gradually become historical-geological individuals which have proper names ("Wadden Sea," "Schlei," "Jasmund," "Darß"). The predicament of particularity is entailed in the famous slogan: "Think global, act local." Marine policies often also operate on national, or even federal and communal scales. Lieven (2020) has argued that national states and political entities as EU are indispensable for environmental policies. In any case, it makes good sense to create as many refugees and recoveries of nature on minor scales, hoping for beneficial up-scaling effects in the longer run.

Therefore, comprehensive studies on marine and coastal areas of smaller seas are needed. This is the aim of our study. Many general human impacts on marine systems are actual on smaller scales, as at the Baltic Sea. If so, the Baltic Sea can be seen as a "laboratory" for challenge and response, pressures, and outlooks for recovery in terms of ecosystem services at coastal zones. In 2004, the German Environmental Advisory Council published a report on marine environmental protection for the Northern Sea and the Baltic Sea (SRU 2004). This report relies on many HELCOM and OSPAR reports and documentations (see also WBGU 2013; MARE 2017).

A focus on ecosystem services must integrate scientific and social science perspectives as the ecosystem service approach wishes to bridge the gap between nature and human welfare. The approach as such requires inter- and even transdisciplinarity. Natural sciences can identify pressures on ecosystems, as on marine and coastal systems. Science can detect causalities, model complex interactions, and, with some caution, predict outcomes. Natural sciences can observe how ecosystem services change over time. Social sciences deal with dispositions of human behavior (psychology), opportunity costs (economics), institutions (law, political science), social stratification (sociology), and inclinations to react on incentives (behavioral economics). Social sciences are about empirical societal affairs, seen as matters of fact ("soziale Tatsachen"). Empirical sociology investigates how people factually value ecological services. They also may point to challenges, to which societies *should* be able to respond. In the sphere of academia, there are some normative disciplines as well: ethics, political philosophy, economics, and legal studies, which have some expertise in how matters should and should not be.

Ethics is about how people *should* (not) behave. Ethics wishes to substantiate moral yardsticks, as principles, ideals, and virtues. Economics as well has some prescriptive content as it is about "rational choice" and "efficiency." It can be based in an anthropology of human dispositions. Legal studies, which make some suggestions “de lege ferenda,” also belong to the scope of normative disciplines. Disciplines, which gather around the flag of “sustainability science” often, combine scientific and normative components (Ziegler and Ott 2011). Since our study belongs to this type of trans- and interdisciplinary inquiry including normativity, some reflective remarks are appropriate.

Since decades, there is a growing demand for interdisciplinary and transdisciplinary (some say: “post normal”) science. Inter- and transdisciplinary modes of research are driven by real-world problems. Interdisciplinary science addresses problems not just from different scientific lenses, but wishes to integrate them. A close reflection on interdisciplinary validity claims is given in Gethmann et al. (2015). Transdisciplinary sciences involve lay persons, professional stakeholder, persons from administrative bodies, and local authorities. Such inquiries assume that scientific knowledge is only one body of knowledge among other ways of knowing. Thus, local, indigenous, and professional knowledge should be incorporated into problem perception and solution. Pohl et al. (2017) make a ten-step proposal of how to perform transdisciplinary science successfully.

Inter- and transdisciplinary approaches, however, presuppose disciplinary excellence. The contribution of each discipline should rest on high scientific standards. Other requirements are the separation of facts and values, the difference between predictions and scenarios, definition of the limits of science (uncertainty, unknown unknowns), and transparency of evaluation schemes (as Ecosystem Service Approach), and policy suggestions.

Environmental studies are a paradigm case for inter- and transdisciplinary inquiries, which include empirical and normative disciplines. In such studies, both epistemological and normative reflections are needed. If values and obligations are made fully explicit in discursive ways, scientific disciplines can be engaged in societal transformations, be it sustainability science, conservation biology, restoration ecology, environmental law, marine conservation, etc. Thus, the distinction between the “two cultures” (Snow 1959) of science and humanities should be complemented by a “third culture” of interdisciplinary environmental sciences (Ziegler and Ott 2011; Ott 2014). This investigation on the Southern Baltic Coast has been written presumptively out of the spirit of such “third culture.”

Our inquiry rests on the commonly shared normative assumption that environmental degradation and the loss of ecosystem services should count a paramount challenge. Thus, we wish to bring together scholars from different sciences and we have to include environmental evaluations (for a philosophy of environmental evaluation see Ott and Reinmuth 2021). Comprehensive environmental evaluation can make use of different schemes, as Total Economic Value, Ecosystem Service Approach, and the universe of environmental ethical discourse (for overview see Ott 2020). In environmental evaluation, there is an interplay between sociology and ethics. While sociology informs how individuals or groups values ecosystem services, environmental ethics takes a normative perspective. We will deepen the environmental ethical dimension in Chap. 19.

2.2 Individual and Collective Demands for Marine Ecosystem Performance

People's well-being and life satisfaction ultimately depend on the extent to which their needs, aspirations, and desires, as well as the motivations underlying these, are fulfilled. Research has long sought to address these relationships. To do so, the field has primarily relied on work in motivational psychology, which considers the extent to which human behavior aims to fulfill needs and desires, and how these needs and desires arise.

As noted above, ensuring well-being is a central human need. Accordingly, people seek to bring about positive, emotionally beneficial experiences, and at the same time to avoid or prevent negative emotional experiences. Human behavior can therefore be said to be motivation-driven (Kroeber-Riel and Gröppel-Klein 2019, p. 157). It has been shown that motivations shape our behavior—both consciously and unconsciously.

Research in motivational psychology has primarily focused on the idea that individually distinct drivers can be understood in light of a small number of basic motives. For example, Rothermund and Eder (2011, p. 95 f.) stated that all human behavior can be traced back to and divided into three basic motives: striving for power, will to power, and desire for connection or attachment. The best-known classification of motivations is Maslow's (1975) hierarchy of needs. This hierarchy is based on three premises (Maslow 1970):

- All humans have a similar set of motivations.
- Some motivations are more basic or critical than others.
- The more basic motivations must be satisfied to a minimum level before other motivations become relevant.

Based on these premises, Maslow proposed his hierarchy of needs according to the pyramid shown in Fig. 2.1.

As shown in Fig. 2.1, physiological needs must be satisfied to a certain (individual) level before safety needs become relevant. In turn, once safety needs have reached a satisfactory level, belongingness, esteem, and self-actualization become dominant motivations (Schiffman and Kanuk 2007).

At a societal level, these individual motives and needs aggregate into collective needs. The basic (physiological and safety) needs have high priority in society; however, the satisfaction of higher, more hedonistic needs contributes significantly to life satisfaction of societies as a whole.

Against this background, many countries have sought to anchor collective needs in their state goals, often in a financial sense via growth targets for their gross national product, but occasionally via indicators such as the Gross National Happiness Index, in which the life satisfaction of society is determined not only financially, but also from human and psychological perspectives.

The satisfaction of the above-described motives and needs is influenced—at both individual and collective levels—by marine ecosystems. In the literature, services

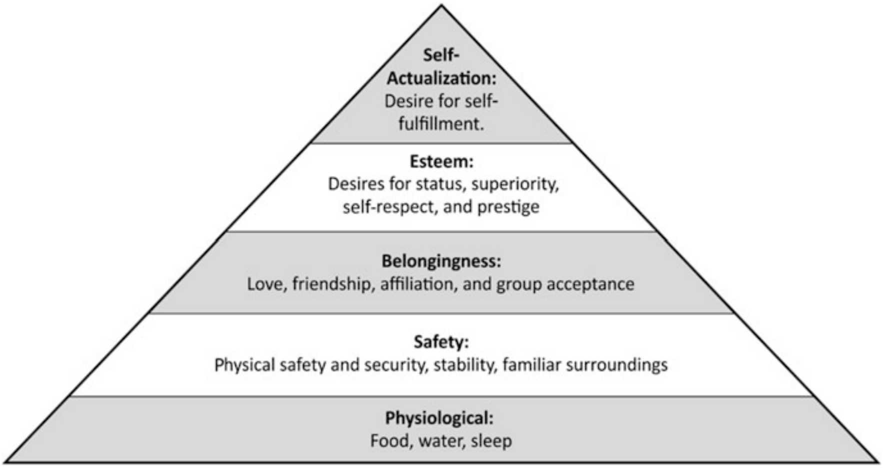

Fig. 2.1 Maslow's hierarchy of needs and ESS gratifying these needs (Mothersbaugh and Hawkins 2016)

related to these ecosystems have been divided into regulating, provisioning, and cultural services (Hernandez-Blanco and Costanza 2019). Regulating services of coastal maritime systems creates value by means of, for instance, flood control or climate control and in this way gratifies safety needs. Provisioning services of coastal ecosystems provide human beings with seafood or costal specific plants like seaberries and gratify physiological needs. Finally, cultural services yield spiritual, recreational, and aesthetic benefits and therefore gratify belongingness, esteem, and self-actualization.

Thus, maritime ecosystem services on both the individual and the collective levels contribute to the life satisfaction of both individuals and society. Regulating and provisioning services—according to Maslow's subdivision—are geared more towards physiological and safety needs, while cultural services primarily address hedonistic needs.

However, it remains largely unexplored as to how the services of maritime systems are created, and how the biological processes work together to produce regulating, provisioning, and cultural services. Likewise, there remains a gap in the literature regarding the interactions between marine ecosystem services and the life satisfaction of individuals and entire societies measured using gross national product or happiness indices.

Against this background, a wide variety of scientific disciplines, e.g., ethnographic research methods embedded in transdisciplinary assessment approaches must be drawn upon to investigate the interdependencies with respect to both the origin and the impact sides of marine ecosystem services. In particular, there is a need for interdisciplinary research between natural sciences, economics, social sciences, and psychology. The current study attempts to provide such an integrative, interdisciplinary view of marine ecosystem services.

2.3 Disciplinary Starting Points of the Analysis

2.3.1 Aspects of Marine Ecology

Compared to coastal, and even more drastically, terrestrial ecology, marine ecology until now can be characterized as a field where a few scientists, equipped with very expensive instrumentation and requiring huge resources try to get at least a glimpse into structure and function of the ecosystems occupying 71% of earth surface (e.g., Odum 1999). However, throughout the past decades, it became clear that not only coastal, but also offshore ecosystems are being drastically influenced and altered by human activity (e.g., Pauly et al. 1998; Roberts 2007), raising the first reports of signs for this (e.g., Hempel 1977) to a kind of global certainty.

Especially for the Baltic Sea, one of the few larger marine systems studied with reasonable temporal and spatial resolution (e.g., Kautsky and Snoeijs 2004) human impact has been shown to alter structure as well as functioning of the offshore coastal ecosystems (Österblom et al. 2007; Korpinen et al. 2011; Andersen et al. 2015). Main reasons for this are seen in an increased human population in the catchment area, raising from about 14.4 million around 1700 to present-day ~85 million (Zillén and Conley 2010) and in parallel increased intensity of direct (e.g., transportation, pollution, eutrophication) as well as indirect (e.g., temperature regime and saltwater inflow) anthropogenic pressures (Laamanen et al. 2017). Consequences from these alterations of ecosystem structure and function are numerous, most prominent probably the two regime shifts identified by Möllmann et al. (2008), which took place in the late 1980s and mid 1990s, respectively.

However, having a rather good temporal and spatial resolution of data, if compared to marine ecosystems in general, human pressure and impact differs largely between the regions of the Baltic Sea. As clearly shown by the analysis published by HELCOM (2010a), pressure and impact is highest for the southern Baltic, Gulf of Riga and Gulf of Finland, whereas Bothnian Sea and Bothnian Bay are less impacted. Consequently, studying the effects of human impacts in areas with high pressure was in demand and a comprehensive overview about biological, hydrological, and climatic changes during the past decades, focusing on the southern Baltic Sea, was published by Feistel et al. (2008).

Extending this extensive baseline data for the past years (Chaps. 9–18) and filling some gaps with respect to, e.g., bioturbation (e.g., Chap. 15) created a sound background for the development of scenarios to be evaluated for their ecosystem service distribution patterns. This evaluation was the first time marine ecology aspects became analyzed comprehensively in the context of ecosystem service provision, reflecting the full spectrum of feedback mechanisms between alterations of ecosystem structure and function due to human activities and consequences for societal welfare.

2.3.2 Aspects of Costal Ecology

Coastal ecosystems worldwide are receiving immense anthropogenic impact, altering both, their structure as well as their functioning (e.g., Doney 2010). More than a billion people are estimated to live in low-lying coastal regions (IPCC 2007) and both, increasing coastal population numbers (Kummu et al. 2016) as well as intensified human activities, ranging from expansion of settlements via construction of coastal protection measures, harbors, and wind farms to tourism (Nicholls et al. 2007) left their marks on coastal ecosystems. This applies especially for the Baltic Sea, surrounded by industrialized countries with a total population of over 70 million people in their catchment area. Strong eutrophication, reflected in the accumulation of high amounts of organic matter and detritus in the sediments and in the water column (e.g., Schiewer 1994; Meyer-Reil 1999) resulted in the need of concerted action combating decline in ecosystem health (e.g., HELCOM 2010b). As a result, decreasing levels of coastal loads could be observed already in the late 1990s (Bachor 1996) due to the development of local sewage plants and reduction of fertilization. Consequently, southern Baltic coastal ecosystems, irrespective of staying still one of the most productive areas of the Baltic Sea (Schiewer 1998), altered their structure and function once again but without returning to their pre-industrial status (Schubert et al. 2010).

Being studied intensively for more than five decades (Schiewer 2008), causal analysis of the structural and functional changes observed in Baltic coastal ecosystems should be possible. But in practice, it becomes a challenge because:

1. The coastline creates a unique transition zone between terrestrial and aquatic ecosystems, being treated by different disciplines which rarely included this ecotone in their investigations.
2. The brackish conditions of the Baltic Sea result in peculiarities with respect to the available species inventory (e.g., Remane 1934; Schubert et al. 2011; Telesh et al. 2013) as well as sensitivity to abiotic variability (Pilkataityte et al. 2004; Telesh et al. 2021), and.
3. Most detailed investigations started in the 1970s, when systems were already largely altered (Schubert et al. 2010; Sagert et al. 2008; Selig et al. 2006).

Whereas the first challenge could be addressed by combined efforts of terrestrial and aquatic ecologists, tightly working together by gathering the database for this book and a number of recent investigations helped by addressing the second one, the third challenge remained an obstacle.

Historical mapping approaches as well as herbarium records indicate a strong decrease of macrophytobethos until about the mid-1990s followed by a recovery (Yousef and Schubert 2001; Schubert and Schories 2008) but still exhibiting large interannual fluctuations (Selig et al. 2009). Most of the detailed synoptic investigations of structure and function of Southern Baltic ecosystems started when macrophytes-dominated systems were already scarce and seen as “marginal.” The few studies available from previous periods (e.g., Schnese 1980) do not include benthic-pelagic interactions as well as sediment dynamics (Schiewer 1998). Both elements have been proven to be crucial for stability and resilience of macrophyte-dominated freshwater ecosystems (Vollenweider 1977; van Nes et al. 2003). On the other hand, attempts to transfer these results to brackish conditions failed (Jeppesen et al. 1998), leaving a big question mark with regard to the role macrophytes stands play for matter fluxes and appearance of coastal ecosystems.

Besides pure academic interest, this gap of knowledge also touches several applied aspects. Restoration measures are costly, and spending societal resources requires sound justification. For this, economic aspects as, e.g., value of clear water for tourism, impact of macrophytes stands for fish recruitment, etc., must be evaluated the same way as aspects of ethics, culture, and consequences for neighboured ecosystems (see Chaps. 19–26).

2.3.3 Aspects of Ecosystem Analysis

Ecosystem analysis and ecosystem research try to investigate the interrelations between the structural and functional entities of spatio-temporal ecological units in order to provide integrative and functional information for holistic decision-making processes. This research direction is a relatively young branch, basing upon the first mentioning of the term "ecosystem" by the botanist Tansley in 1935. Already in his very first definition, Tansley (1935, p. 299) stressed the integrative character of the approach, underlining that *"the more fundamental conception is, . . ., the whole system (in the sense of physics), including not only the organism-complex, but also the whole complex of physical factors forming what we call the environment of the biome. . . . It is the systems so formed which, from the point of view of the ecologist, are the basic units of nature on the face of the earth. These ecosystems, as we may call them, are of the most various kinds and sizes."* During the following years, these holistic ideas were developing slowly, reaching one innovative summit after a linkage with cybernetics and ecological modelling in the 60s (Patten and Joergensen 1995) and several applications of energy flow analysis (e.g., Odum 1971, 1983). This tradition of joining the concepts of ecology and systems analysis was deepened thereafter and has resulted in extensive applications of IT tools and models (Joergensen and Fath 2011). Besides modelling and practical analysis, also different ecosystem theories were developed, which focussed, e.g., on thermodynamics (Nielsen et al. 2020; Joergensen 1997; Joergensen et al. 2011), network theory (Patten 1991), information theory (Ulanowicz 1986), hierarchical approaches (Müller 1992), resilience analysis (Holling 1986), or gradient dynamics (Müller 1998). Also the empirical ecosystem research was rising since the 80s with comprehensive interdisciplinary projects (e.g., Likens et al. 1977; Tenhunen et al. 2001; Fränzle et al. 2008). The rising knowledge was used in environmental management and planning (Haber 1994), forest dieback (Ulrich 1990), and coastal management (Ecosystem research projects in the German Wadden Seas, Behrends et al. 2004, the Baltic Sea Wulf et al. 2001a, b). As a consequence of the sustainability paradigm, the scope was opened further after 2000, the subject became the human-environmental system and the connection to economy and sociology was aimed to be closed by the ecosystem service approach (Grunewald and Bastian 2015; Chicaro et al. 2015).

The ecosystem-based, holistic investigations have been crystallization nuclei for environmental protection concepts, such as ecosystem health (Rapport et al. 1998; Costanza 2012), ecosystem integrity (Woodley and Kay 1993; Müller 2005). They have levelled a path for the CBD ecosystem approach and the application of the DPSIR indicator analysis of the European Environmental Agency (e.g., Hou et al. 2014; Patricio et al. 2016). Meanwhile, several applications of the ecosystem approach can be found in several programs for environmental or climate management, be it the valuation of ecosystem states by ecosystem services and ecosystem conditions (Roche and Campagne 2017), or the basic holistic fundamental of the European Water Framework Directive or the *Marine Strategy* Framework Directive (Lillebö et al. 2016) of the EU. Therefore, we can find several applications and progressions in BONUS and HELCOM programs, in the concepts of marine spatial planning (Douvere and Ehler 2009; Schernewski et al. 2018), in the practical assessments of wind farms, the management of eutrophication processes, within national and international projects.

This is also the case referring to several KÜNO projects (see Chap. 1) among them BACOSA and SECOS, which are the subjects of this publication. In both cases, the structures, functions, and organizations of marine ecosystems around the German Baltic coast have been analyzed with respect to spatial distribution patterns, temporal developments, and in-depth analyses of energy, nutrient, and information flow through the biotic-abiotic networks. The attempt was made to follow some physiological schemes of ecosystem service production in order to link the ecological analyses with the sustainability framework conditions of human welfare.

2.3.4 Aspects of Environmental Economics

Economics deals with the allocation of scarce means to competing ends. In striving for their individual well-being and life satisfaction, humans use scarce resources, which in many cases are allocated via markets. If markets are perfect, the resulting allocation of resources is Pareto optimal. No one can be made better off without making someone else worse off. The market solves use conflicts efficiently by allocating scarce goods to the agents with the highest willingness to pay for them. However, markets are rarely perfect and a major imperfection is the existence of externalities. An externality is a positive or negative impact of an activity on another agent's well-being with no compensation being paid. In the case of a positive impact, the activity level is too high since the agent can reap only a part of the total benefit; in the case of a negative externality the agent bears only a part of the cost and her/his activity level is too high. It is obvious that externalities are omnipresent in situations when activities like production or consumption have environmental impacts. A major role of environmental economics, therefore, is to identify and measure

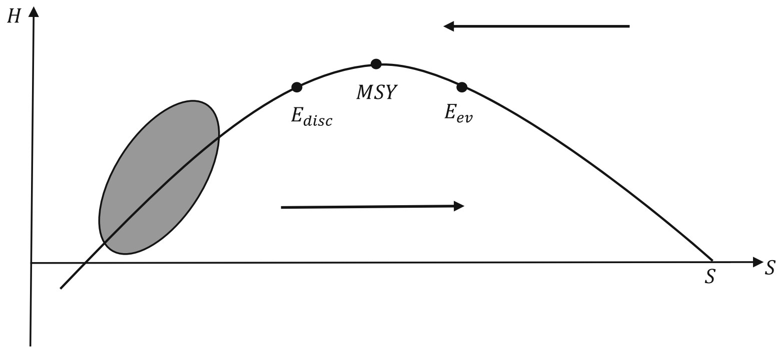

Fig. 2.2 The regeneration and use of a marine resource

environmental externalities and to suggest ways in which the resulting market failures may be corrected.

Use conflicts regarding environmental resources involving externalities are omnipresent and they are particularly important in coastal areas. Given that 40% of the global population live within 100 km of the coast, humans exert substantial pressure on offshore and onshore ecosystems. Use conflicts include the use of the ocean as a receptacle for pollutants such as sewage or fertilizer runoffs versus its function as a public good for consumption (e.g., tourism) or production (e.g., fishery) as well as competing uses of scarce land in coastal areas for production, human habitation, ecosystem services and other activities. Decisions on the use of coastal and maritime resources often have long-term effects or are even irreversible. Many harmful substances discharged into the environment are persistent pollutants, decaying only slowly and causing damage for many years or even decades. Moreover, the exploitation of fish at rates beyond their natural regeneration reduces future catches and may result in the collapse of fisheries like one experienced in Newfoundland in the early 1990s. Möllmann et al. (2021) argue that the Western Baltic cod fishery is very close to a similar tipping point. These intertemporal trade-offs must be considered when decisions about the use of environmental resources are made and questions regarding sustainable development and intergenerational equity enter the picture.

The intertemporal and intergenerational dimensions of resource have been modelled by economists in dynamic models of resource use. See Perrings (2016) for an overview and Clark (1990) for a more comprehensive treatise. The simplest model is the standard renewable resources model, which may be interpreted as a single-species fishery, and it will be used for illustrative purposes here. Extensions such as multispecies models or models with different age cohorts are possible and have been discussed in the literature—even with applications to Baltic fisheries. See Tahvonen et al. (2013) and Bauer et al. (2019), for example.

There is a stock of a resource, S, e.g., an economically relevant fish species like Baltic cod. This stock can change over time and the hump-shaped curve in Fig. 2.2 is the natural regeneration function. The maximum stock $\overline{S}$, which cannot be surpassed, is the ecosystem's carrying capacity for this species. Moreover, there may be a critical stock below which regeneration becomes negative, i.e., the species is getting extinct. The variable on the vertical axis is the harvest rate H. If the harvest is larger than the rate of regeneration, the stock of the resource declines. If it is smaller, the stock grows. This is depicted by arrows that indicate the direction of motion. The simple bioeconomic model may be taken as a metaphor of the more general problem of the economic use of complex dynamic ecosystems with regenerative capacities. Of course complex systems require complex strategies, which cannot be handled within the framework of a one-species textbook model. Nevertheless, the diagram is still useful to illustrate some conceptional issues.

A long-run economic-ecological equilibrium requires that the stock of the resource is constant. i.e., the rate of harvest must equal the rate of regeneration and feasible all long-run equilibria lie on the hump-shaped curve. The maximum sustainable yield is depicted as *MSY*, but economic models have shown that other points can be long-run optima, too, depending on how the resource is evaluated. If future harvesting revenues are discounted, the equilibrium is shifted to the left, E_{disc}, indicating that future generations will have less access to the resource than in the *MSY* scenario. If, on the other hand, the resource has an existence value beyond the simple revenue of harvesting, the equilibrium is shifted to the right, E_{ev}. This becomes relevant if we deal with maritime ecosystems providing valuable ecosystem services beyond their utilization for commercial fishing. The shaded area characterizes a situation of over-exploitation of the resource with the possibility of extinction. See Fig. 1 in Quaas et al. (2018), where a similar diagram is used to visualize the decline of Eastern Baltic cod stocks.

Over-exploitation is the result of externalities and open access and Gordon (1954) is the seminal contribution to the literature. In a benchmark scenario with open access, fishers maximize profits by comparing the price in the market to the marginal cost of the fishing effort like fuel cost, wages of the crew, their own time, etc. However, they do not consider, the intertemporal effect of the reduction of the stock on future fishing opportunities. This cost is borne by society as a whole (including future generations) and constitutes a negative externality. Economists have coined the term "social cost" to distinguish the impact on society from the private cost borne by the individual agent. As negative externalities induce excessive harvesting, overfishing and possibly the collapse of the fishery are direct consequences of open access to the scarce resource.[3] This is Hardin's (1968) "Tragedy of the Commons." The environmental economist tries to solve the externality problem by (1) determining the socially optimal solution, which takes account of the full cost of resource use, and (2) by suggesting environmental policies that make individuals behave such that the scarcity of the resource is incorporated correctly. The policy establishing such a solution involves a system of fishing quotas with a cap ensuring

[3] Exhaustion can be avoided if the private cost of fishing increases drastically as the stock declines, such that the harvest goes to zero before the stock reaches its critical lower level. In Gordon (1954) this is modelled via the Sheaffer catch function, which has exactly this property.

that unsustainable harvesting rates are excluded or, alternatively the taxation of fishing effort such that the catch is reduced from its open access to the socially optimal level. The tax rate establishing the optimum is called the Pigouvian tax after Arthur Cecil Pigou, who invented this instrument as a means to internalize externalities and to signal the social cost to the individual user of the resource, Pigou (1920). Environmental economists have shown that the Pigouvian tax and an ideal cap-and-trade system of (fishing) quotas produce the same result. A major advantage of these market-based instruments is that they provide efficiency. High-cost firms have an incentive to sell their quotas in the market and low-cost firms have an incentive to buy and use them. Likewise, only low-cost firms are willing to pay the Pigouvian tax, which gives them access to the resource. Thus, the resource will be harvested by the firms with the lowest costs. Although taxation and cap-and-trade systems have gained importance in practice, in particularly in clean-air and climate policies, they are not yet very prominent in the regulation of maritime resources. In the case of the Baltic, fisheries have been regulated mainly on command-and-control basis with instruments such as total allowed catches (TACs), regulations on bycatches and mesh sizes, and the temporal or spatial closures of certain fisheries (ICES 2020). These regulations have contributed to conserving fish stocks in the Baltic although, as mentioned above, some authors argue that situation of the Western Baltic cod is critical.

Matters become even more complicated if the impact of human activity on complex ecosystems instead of relatively simple fisheries is to be analyzed. Consider, for example, the problem of maritime eutrophication, a major issue in the Baltic, which is mainly caused by landside activities, in particular agriculture. The causal chains transforming land-based fertilizer use into maritime eutrophication are complex and hard to be tracked. In environmental economics, the term of nonpoint source pollution has been coined to characterize the phenomenon and policy recommendations to deal with unobservable emissions have been derived. See, e.g., Xapapadeas (1999). The major problem faced by the economist, however, is to determine the “true” marginal value of ecosystem services. As there are intertemporal trade-offs, the parameters connecting the present with the future are decisive. In the ecological system, this parameter is the capability of the ecosystem to regenerate, or in more general terms; its resilience. If regeneration is slow, damages to the ecosystem can easily become irreversible. Thus, future ecosystem services are scarce and dear and in consequence, environmental regulation must be stringent. On the economic side, the present and the future are connected by the discount rate. If future benefits are discounted at a high rate, losses of ecosystem services that affect future generations do not really matter and regulation can be lax. The third and possibly most decisive parameter, however, is the monetary value of ecosystem services, i.e., the loss society would suffer from a deterioration in ecosystem quality. All three parameters are hard to be determined. The regenerational capabilities of complex ecosystems are difficult to assess and sometimes become obvious only after irreversible changes have occurred. The choice of the discount rate is a delicate ethical question of valuing and comparing the well-being of the present vs. future generations (see Chap. 19). Finally, measuring the benefits society derives from ecosystem services is at least as difficult as determining ecosystem resilience quantitatively. Economists have developed methods to assess the value of non-market goods using various approaches to determine the willingness to pay for these services. There are major problems with this, e.g., the fact that willingness to pay is always limited by ability to pay such that rich people count more than poor people when ecosystems are evaluated or the problem that "stated preferences" may deviate systematically from the true willingness to pay. Moreover, a large share of the value is an "option value," related to hitherto unknown benefits and opportunities that might materialize in the future. Nevertheless, economic evaluation of environmental resources is an important tool for environmental policy advice. Finally, economic research supports policy advice by suggesting ways in the which sustainability goals agreed upon by the involved parties can be achieved efficiently, i.e., at the lowest possible cost.

2.3.5 Aspects of Environmental Ethics: A Philosophical Rationale for Ecosystem Services

There are even some arguments stemming from theoretical sociology why the ecosystem service approach (ESS) should serve as conceptual device for environmental evaluation at the Southern Baltic Sea. Since the eighteenth century, European cultures and states move into modern societies. Such societies are not integrated in a hierarchical "top-down" manner any more, but they differentiate themselves according to specific social functions. The process of modernity establishes functional systems as law, economy, education, science, art, religion, media, private life, and politics (Luhmann 1984). As Luhmann (1986) argues, nature is not well represented by the generic codes of societal systems, as "truth" in science, "money" in economics, the "sacred" in religion, and "governmental power" in politics. These generic codes function as lenses by which a social system can "register" (modified) nature. Nature is not a social system, but it is rather the environment of all societal systems. The systemic lenses present nature in specific modes, as extraction costs (economy), property rights over land (law), convictions of new protest movements (politics), inspiration for works of art (art), empirical objectivity (data-based science), forces and chemistry (technology), etc.

System theory, however, distinguishes the layers of (a) codes and (b) programs. Codes are fixed, programs are flexible and open for reforms. The many programs within single social systems can be shaped and re-designed according to environmental challenges. In this respect, the ecosystem service approach can be of paramount help. The ecosystem service approach makes good sense in different systemic programs (law, economics, psychology, geography, ethics). It can serve as a commonly shared terminology assisting communication as well between single academic disciplines (interdisciplinarity) and between academia and other societal systems (transdisciplinarity).

System theory, however, has a blind spot: the *lifeworld* (Habermas 1981). The concept of lifeworld refers to an inexhaustible network of believes (values, convictions) which remains mostly in the background of agents but provides agents with patterns of meaning which are underrepresented by the logic of functional systems (and also by the logic of pure science). The values of nature are stored within the deep background of the lifeworld. Individuals can explore such values in interactions with nature, as in hiking, riding horses, gardening, diving, sailing, bird watching, and even hunting. Such practices by which humans encounter valuable nature are important structural connectivities between natural and human systems. Sharpening the systemic lenses and mobilizing the lifeworld as storage pit of the multitude of nature's values are two complementary strategies for nature conservation as seen from a theory of society. The lifeworld entails the dimension of cultural values, commitments, and personal attitudes. Thus, modern societies are always within the dialectical interplay between functional systems (functions, codes, programs) and lifeworld (practices, values, attitudes, commitments) (Habermas 1981). The ecosystem service approach is designed to make visible natural benefits within economic, societal, legal discourse, and decision-making. As a unified evaluation scheme, it can bridge the gap between single social systems and play a constructive role in different programs. It is of relevance both for social systems and for components of the lifeworld.

Thus, ecosystem service approach cannot just bridge gap between single social systems, but also between systems, the lifeworld and even the spheres of normative orders (law, public administration, international regimes, EU-directives, etc.). These conceptual advantages of ESS might, perhaps, turn into shortcomings at the global scale where many non-Western modes of life and interactions with nature must be accounted for on their own cultural terms. On global scales, ESS might be accused of being biased in favor of Western modes of thought. The debates within IPBES have reflected this contested topic (see e.g., Diaz et al. 2015; IPBES 2019). Since the Baltic Sea is a European region at a minor scale, we can be rather relaxed about this topic. A marine region surrounded by modern states of the European Union (and Russia) can reasonably be assessed by ESS. Under the ESS-approach we can and should point at the importance of cultural services. Since spiritual values are included in cultural services, we can even make room for spiritual encounters with nature, be they Christian or be they rooted in pagan traditions which survived at the peripheries of Baltic regions. Thus, there is a theoretical social science reason for a conceptual choice in favor of ecosystem service approach with respect to our study area.

2.4 Interdisciplinary Structure of the Book and Detailed Research Questions

Throughout the previous text passages, some of the involved scientific directions have been described briefly, focussing on their specific aspects with respect to coastal ecosystem analyses. The motivation for a multidisciplinary analysis has been found in social-ecological demands and in the need for interdisciplinary conceptions. For instance, we have dealt with marine ecology, coastal ecology, ecosystem analysis, environmental economics, and environmental ethics. It has become visible that disciplinary starting points and targets for analyzing feedback mechanisms between human society and ecosystems differ largely. We might use eutrophication as an example: The origins of nutrient loads mostly stem from the terrestrial hinterland, they are consequences of human population concentrations, human land use, or agricultural activities. Thus, their initial components, their drivers, and their respective motivations are related to economic or social structures. By environmental transfer processes, the nutrients reach coastal ecosystems, where their fates, impacts, pressures, and process rates are investigated with (nature)-scientific approaches. In the end, there are retroactive effects from ecology to society, which might hamper ecosystem service production or produce risk for the human population. Here we are back again in the middle of social sciences.

Within this circle, the natural sciences mainly focus on understanding the effects caused by human impacts with respect to functional and structural changes. On the other side, the humanities are mostly status-oriented with little interest in the analysis of natural laws or pattern creation. Ecosystem service assessment is therefore by no means “putting pieces together.” It is more an interactive learning process, where understanding the motivation and methods of the respective disciplines allows to reflect the limits of knowledge of the own discipline with respect to the driving “societal demand.” Consequently, the anthropogenic approach has to be accepted by natural scientists as well as laws of nature by humanities. Especially if sustainable development is applied as a target function, there is no way to any solution of the comprehensive socio-ecological problems without interdisciplinary cooperation.

This book will guide the reader—irrespective of where he comes from—through such a process, setting the frame and explaining state of the art, as achieved by the intense efforts of the past years, in a disciplinary way. However, we will also highlight the links to the other disciplines in the next partner chapters, explaining the instruments and the developments made by combined research. The structure of the following Chapters therefore starts by describing the constraints mainly from the scientific side—the environmental conditions—in Chaps. 3–8. Thereafter, the actual ecological processes and their environmental interrelation are described in Chap. 9–18. Finally, in Chap. 19, the human side will enter the scene. Moreover, after a description of the efficient human factors from an analytical viewpoint, the approaches are integrated by observing some significant ecosystem service conditions within the study area (Chap. 20–26). In Chaps. 27–30, the human-environmental systems are interpreted and discussed from transdisciplinary viewpoints, and in Chap. 31, some focal conclusions will be drawn.

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Part II

Ecological Structures and Functions of the German Baltic Sea Coast

The Abiotic Background: Climatic, Hydrological, and Geological Conditions of the Southern Baltic

3

Hendrik Schubert, Sabine Bicking, and Felix Müller

Abstract

This part provides a general description of the abiotic conditions of the Baltic Sea, which gives the reader the basic background required for understanding the peculiarities of the system investigated. Being a microtidal brackish water system, large with respect to area and volume, but having just narrow connection to the adjacent ocean, pronounced, but relatively stable gradients in salinity are a unique feature making the Baltic something special. In addition, post-glacial history resulted in characteristic patterns of subsoil geology as well as prevailing coastal types, which are presented on a Baltic scale here before being treated in detail for the investigation area.

The Baltic Sea is a marginal sea of the Atlantic, characterized by strong vertical and horizontal gradients with respect to climatic and hydrological conditions. Stretching over 1.200 km in East-West and 1.300 km in North-South direction, it covers an area of 412.560 km^2 (Table 3.1). The total water volume is approximately 21,631 km^3, being replaced within approximately 30 years[1] by the combined effect of salt water intrusions (approximately 4000 m^3 s^{-1}, Schiewer 2007a), freshwater inflow

[1] Just adding up the terms would suggest an exchange time of 32 years but different approaches give turnover times between 26 and 35 years (Meier 2005).

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Table 3.1 The Baltic Sea catchment in numbers (all are rounded approximations)

Topographic features		Hydrological features	
Area	412,560 km^2	River water supply	440 $km^3\ y^{-1}$
Volume	21,631 km^3	Precipitation	225 $km^3\ y^{-1}$
South-North extension	1300 km	Evaporation	185 $km^3\ y^{-1}$
West-east extension	1000 km	Freshwater balance	480 $km^3\ y^{-1}$
Average depth	52 m	Residence time	25 years
Maximum depth	459 m		

Source: Internet page of the Leibniz-Institut für Ostseeforschung Warnemünde (https://www.io-warnemuende.de/steckbrief-der-ostsee.html)

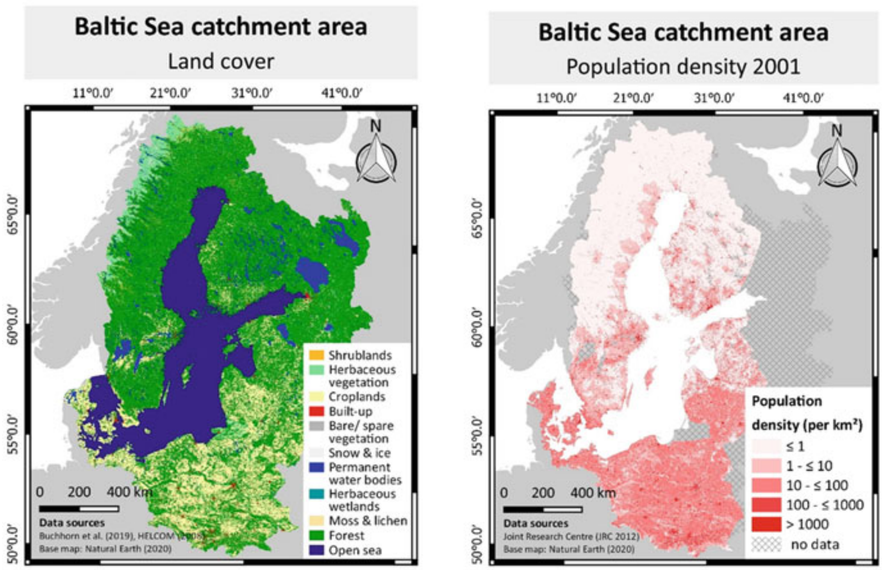

Fig. 3.1 Land use (left panel) and population density (right panel) of the Baltic Sea catchment area

(approximately 16,000 $m^3\ s^{-1}$, Schiewer 2007a), precipitation and evaporation (Table 3.1).

The entire catchment area of the Baltic Sea covers 2.13 Million km^2, i.e., almost 20% of the land area of Europe, distributed over 14 large river basins.[2] Eighty-five million people live in this area. The main land cover classes of the watershed are forests (around 52%), open sea (around 19%), croplands (around 17%) and permanent water bodies (around 5%).[3]

However, there is a strong gradient with respect to land cover class distribution as well as population density (Fig. 3.1). Agriculture (and population) provides the

[2] https://www.baltex-research.eu/background/catchment.html

[3] https://land.copernicus.eu/pan-european/corine-land-cover

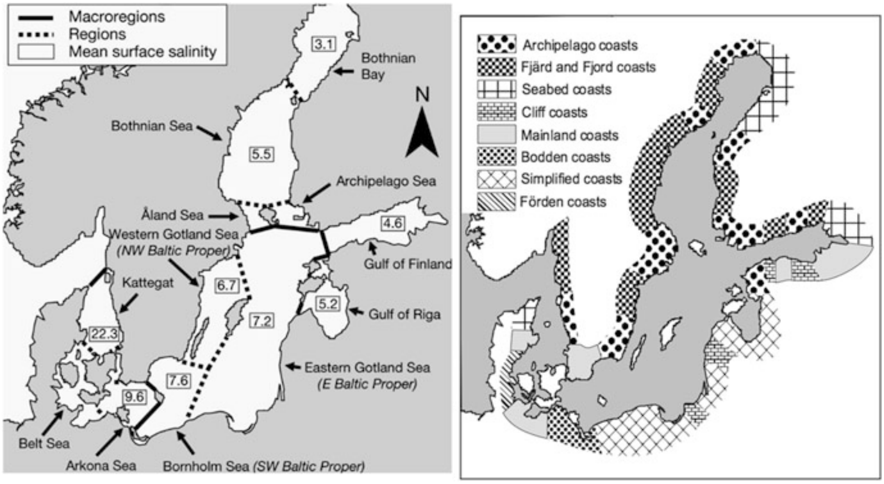

Fig. 3.2 Hydrological regions (left panel) and coastal type distribution of the Baltic Sea. Left panel from Telesh et al. 2011a, right panel from Schubert et al. 2004

dominating land use in the western and southern parts, the focus area of the studies presented here (Fig. 3.1, left panel).

This regional distribution is only in part due to the climatic conditions, becoming more favorable for agriculture in the south, but it also reflects the geological background. Whereas the northern part is mainly formed by Precambrian and Palaeozoic crystalline bedrock, glacial, and pre-glacial deposits, rich in calcium, dominate the geological subsoil of the southern regions (Winterhalter et al. 1981).

The Baltic Sea itself can be divided into up to 12 regions, representing the respective basins which are separated by deeps (Fig. 3.2). Figure 3.2 also depicts the pronounced gradient in surface salinity, which is one of the main characteristics of the Baltic Sea with drastic consequences for biodiversity and ecosystem function (see, e.g., Telesh et al. 2011a, b, 2013). Moreover, a strong vertical salinity gradient separates surface water from bottom water at least between Arkona Sea and North Gotland Sea by formation of a halocline at roughly 50 m depth (e.g., Sjöberg 1992). From this it becomes obvious that Kattegat and Belt Sea differ largely from the rest of the Baltic and therefore often are regarded as transition zones from a biogeographical point of view as well as with respect to hydrological conditions (e.g., Snoeijs-Leijonmalm et al. 2017).

Geologically (for a detailed description see Chap. 4), the Baltic Sea is a young system, its post-glacial history started about 12.000 year ago when melting glaciers filled the basin with freshwater. A first connection to the Atlantic Ocean opened 10.000 years ago, at this time via Southern Sweden. Whether or not a second connection to the White Sea also contributed to the strong salinity variations during this period, called "Yoldia Sea," is not clear yet (Schiewer 2007a). In any case, uplift of the Fennoscandian shield, as a result of deglaciation, closed these early connections at about 9.250 BC, creating a freshwater "Ancylus Sea" which existed until 7.100 BC. Then the basin became connected to the Ocean again (this time via the Danish straits), transforming the System to a brackish “Littorina Sea,” which existed until 4.000 BC. A gradual decrease in salinity, called the “Limnea period” followed, replacing the marine index species *Littorina littorea* by the freshwater mussel *Limnea ovata* (Schiewer 2007a). The current state, called “Mya period” and being characterized by the above-mentioned complex brackish water gradients, started at about 1.500 years ago.

As a result of geological background and Fennoscandian post-glacial land uplift, the Baltic Sea exhibits a pronounced North-South gradient with respect to coastal types (Schiewer et al. 2007b, Fig. 3.2). Whereas in the North (Northeast Gulf of Finland, Bothnian Bay and Bothnian Sea and most of Sweden) seafloor and fjord/fjaerd coasts, often accompanied by numerous small islands (archipelago coasts), prevail, the southern part is dominated by eroding moraine material, resulting in a complex pattern of cliffs with sediment deposits and various kinds of inlets between them. That this pattern indeed is mainly driven by uplift shows the example of the southernmost part of Sweden (Skåne), where a moraine-type coast prevails because this is a subduction area and consequently a mainland coast as well as the northern tip of Jutland, being a seafloor coast as typical for uplift areas. Uplift of the northern part of the Fennoscandian shield and subduction of the southern part in fact are the main reasons for the pattern of coastal types shown in Fig. 3.2.

The southern part, from about mid-Jutland, is dominated by moraine material exhibiting a characteristic pattern of eroding cliffs and deep (“Förden” or “Fjords”) as well as shallow (“Bugt,” “Vig,” “Noor”) inlets between them. Such shallow inlets become more and more dominant further east (“Bodden,” “Haff,” “Wieck”) being progressively more isolated from the adjacent sea by current-driven accumulation of eroded cliff material. South of Kiel the pattern of cliffs and deposition of eroded material become less complex, long stretches of sandy beaches and large shallow inlets, often almost isolated by sand barriers, prevail the “sediment coast.” East of Rostock cliffs become less frequent, the coasts now dominated by large sand accumulations between them which separate a complex pattern of shallow inlets (Bodden, Haffe) from the adjacent Sea (Bodden Coast). Further east, starting just behind the Polish border, sand accumulation by the water current regime created a mature coast with only a few remaining cliffs between them (e.g., Kaliningrad region).

Another characteristic of the Baltic Sea is the lack of substantial tides. Only the Kattegat, being influenced by the North Sea, exhibits substantial tidal amplitudes, the rest of the Baltic Sea is by definition a microtidal system (see Hayes 1979). Already at the Belt Sea tidal amplitudes drop to ~10 cm, a value only exceeded in the Gulf of Finland for the rest of the Baltic Sea (Leppäranta and Myrberg 2009). However, irregular water level fluctuations driven by meteorological forcing are common in the Baltic structuring the upper littoral zone in geo- and hydrolittoral (or winter and summer beach from a terrestrial biologist’s perspective) in analogy to the marine littoral zonation scheme.

As a consequence of all of the above-mentioned peculiarities, especially the salinity gradient and the young age, the species inventory of the Baltic Sea sensu stricto differs from other brackish water systems at least for benthic macroorganisms (Remane 1934; Schubert et al. 2011; Bleich et al. 2011), but probably also for other groups (Telesh et al. 2011a, b, 2013). Consequently the recent Baltic Sea is often regarded an "unsaturated ecosystem" with respect to species inventory (Schubert and Schories 2008), offering room for natural as well as anthropogenic immigration processes (Leppäkoski et al. 2002; Olenin and Leppeäkoski 1999; Gollasch and Leppäkoski 2007). However, the rate of introduction of new species has increased clearly due to human activities, mainly by opening new invasion corridors and intensified traffic (Olenin 2005). Especially coastal lagoons and inlets of the Baltic are now regarded as "centres of xenodiversity" with altered structure and function of their ecosystems, but also the Baltic Sea as a whole became characterized as "sea of invaders" by (Leppäkoski et al. 2002).

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Geological and Sedimentary Conditions

4

Svenja Papenmeier and Helge W. Arz

Abstract

Many benthic ecological structures and processes are closely related to the sedimentological characteristics of the seafloor. This chapter provides information about the sedimentary conditions of the Baltic Sea. This includes the general sediment distribution of the Baltic Sea in relation to its geological history since the last glaciation as well as high-resolution sediment mapping and regional distribution patterns of geochemical parameters in the surface sediments of the German territorial waters.

On geological time scales, the Baltic Sea is a very young shelf sea with a high variation in morphology and sedimentology. Glacial advances and retreats shaped several basins and various moraine deposits form natural sills dividing the Baltic Sea into 17 subareas (Fig. 4.1). The mean water depth of the Baltic Sea is 52 m with a maximum depth of 459 m in the Landsort-Deep (Northern Gotland Basin) (Köster and Lemke 1996). Generally, the western Baltic Sea is much shallower and the German Baltic Sea, in particular, has a mean depth of 19 m and a maximum depth of 47 m in the Bornholm Basin.

The modern Baltic Sea is a landlocked sea with the only connection to the North Sea through the Danish Straits and the Kattegat/Skagerrak in the West. The latter represents an 80–140 km wide and up to 700 m deep passage between Denmark and Scandinavian Peninsula. The Baltic Sea, however, was not always connected to the North Sea. During the last glaciation in the Weichselian, e.g., this passage and a variable portion of the present Baltic Sea region was covered by the large Fennoscandian ice sheet (Andrén et al. 2011; Björck 1995; Köster 1996). At the

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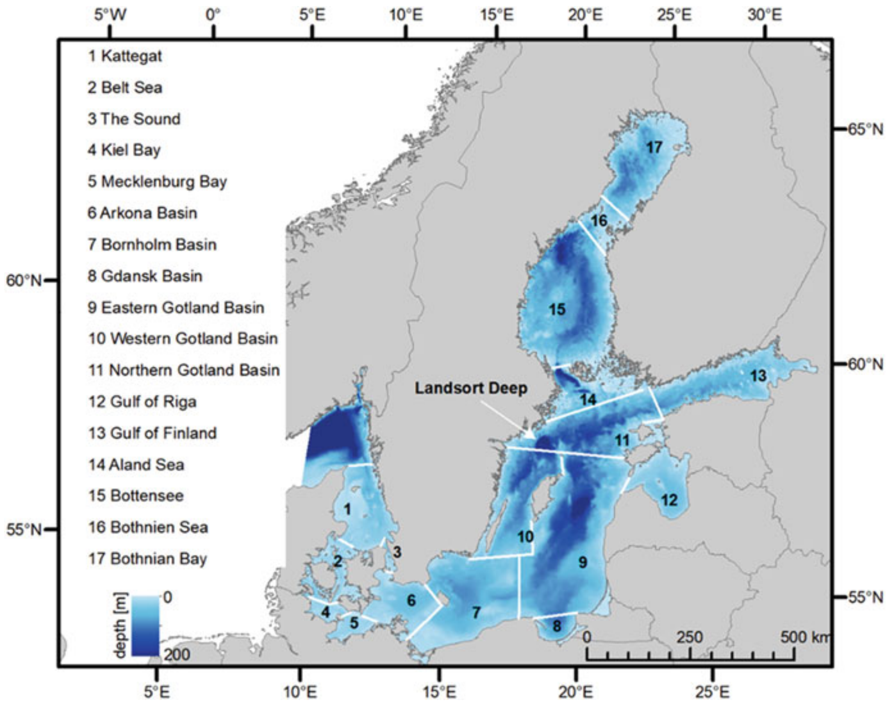

Fig. 4.1 Morphology of the Baltic Sea with its main basins. Bathymetric data set: Seifert et al. (2001)

beginning of the deglaciation about 15 ka BP ago the passage to the North Sea was still blocked and melt water accumulated in the ice-free basins of the southern Baltic Sea. In these proglacial freshwater lake(s) sediment accumulated as clastic varves with bright fine sand layers transported from melt water in summer and dark silty layers deposited during the ice-covered calm winter seasons (Köster and Lemke 1996). By progressive melting, the glaciers retreated further eastward into the Baltic Sea basin and the lake level gradually rose. A first passage to the North Sea developed in central Sweden with the northward retreat of the Scandinavian ice sheet. After a short ~300 a lasting drainage phase of the Baltic Ice Lake (at Mount Billingen; e.g., Mörner 1995) accompanied by a sudden drop in lake level of ~25 m, a subsequent inflow of salt water caused a salty-brackish regime (Yoldia stage, 10–9.25 ka BP) in which more homogenous grayish, silty-clayey sediments accumulated and varve deposits occurred only in small freshwater lakes at the edge of the Yoldia Sea (Köster 1996). The continuing glacio-isostatic uplift of Scandinavia closed this western North Sea connection and melt water as well as drainage water turned the Baltic Sea into a freshwater system again, the so-called Ancylus Lake (9.25–7.1 ka BP). During this time soft silty-clayey rather homogeneous sediments accumulated in the basins (Köster and Lemke 1996). In the marginal regions of the basins wetlands developed that resulted in the deposition of peaty layers. The vast part of the recent western Baltic Sea was still mainland during that time and rivers and lakes characterized the landscape. Parts of the paleo-landscape are still preserved on the present seafloor or detectable with shallow seismics. Examples for this are glacial melt water river systems in the Fehmarn Belt (Feldens and Schwarzer 2012; Tauber 2011), Kadet Channel and Darss Sill, and paleo-forests off Ahrenshoop and Warnemünde (Tauber 2011). The passage to the North Sea was finally opened 7 ka BP when the global sea level reached almost the present level and the brackish Littorina Sea phase started. This stage is divided into two sub-stages Limnea Sea (starting 4 ka BP) and Mya Sea (since 1.5 ka BP) defined by a shift in index species induced by slightly decreased salinities (Köster 1996). During the Littorina phase, the modern coastal landscape developed (Lampe 1996; Schwarzer 1996). Along the predominantly crystalline bedrock coast of Sweden and Finland only minor changes occurred since then. In contrast, major changes occurred at the southern and western Baltic Sea coast. Material accumulated as ground and terminal moraines by quaternary glaciers has been subsequently reworked forming a graded shoreline. In this still ongoing process, the glacial till deposits, with their wide grain size range reaching from clays to boulders, are eroded by wind and waves and form steep, up to 120 m high cliffs. The very coarse fraction of this heterogeneous sediment mixture remains at the base of the cliffs; the sand fraction is transported partly alongshore, while the clay and silt fraction is generally transported offshore to the deeper parts of the basins. Reworking of moraine deposits also takes place offshore. Smaller grain size classes such as silt and clay are washed out, and the remaining coarse fractions (sand to boulders) form so-called lag deposits. The finer fractions together with deteriorated organic matter, originating from the seasonal productivity, are transported to and accumulated in the deeper basins. Here, the subsurface muddy sediments often show a grayish black reduction horizon due to early diagenetic processes and only at the surface a gray-brownish oxidation layer exists. The deepest parts of the basins in the southern Baltic Sea occasionally develop anoxic conditions and organic-rich partly laminated sediments can form.

The Leibniz Institute for Baltic Sea Research Warnemünde (IOW) started in 1992 on behalf of the German Federal Maritime and Hydrographic Agency (in German: Bundesamt für Seeschifffahrt and Hydrographie, BSH), a comprehensive sediment mapping for the German territories. Within this framework, about 2200 granulometric raw data collected since 1952 were analyzed, and results were published on nine sheets (scale 1:100,000) by Tauber (2012a–i). Surface sediments are defined in nine major grain size classes. Additionally, they are each subdivided into five subclasses according to the grain size sorting (Tauber et al. 1999). The comprehensive sediment distribution map shown in Fig. 4.2 overall reflects the interaction of the glacial deposits and the still ongoing postglacial sedimentation processes: sand in shallow-water areas (e.g., Oderbank), coarse sediments (lag deposits) at wave-exposed areas (along the coast and at Adlergrund) and mud in the basins (e.g., Mecklenburg Bay, Arkona Basin). To fulfill, e.g., political demands, a more detailed and area-wide hydroacoustic mapping was initiated by the BSH in 2012. Focus areas are the Natura 2000 sites in the Exclusive Economic Zone (EEZ) of Germany. A mapping guideline (BSH 2016) was developed to enable the

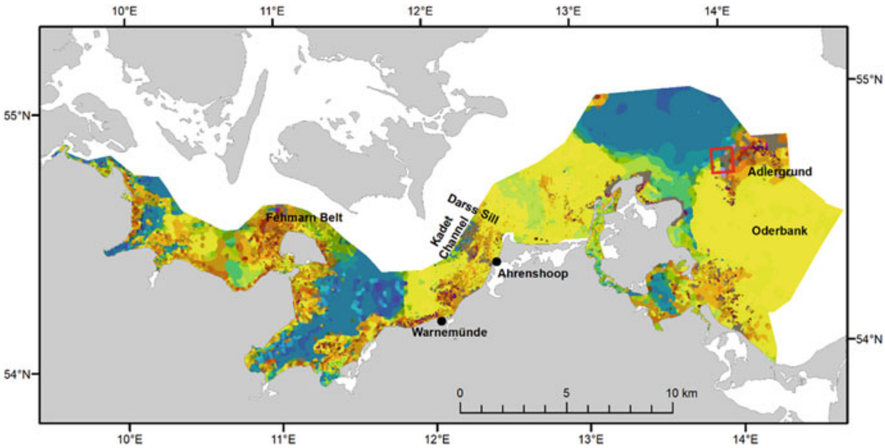

Fig. 4.2 Sediment distribution map for the German territory based on grab sampling and classified after Tauber (2012a–i). Legend see Fig. 4.3. Red box marks the section of Fig. 4.3

objective interpretation of side scan sonar backscatter data with a resolution of 0.25–1 m together with ground truthing data (grab samples, underwater videos). Minimum size of delineated sediment structures is 100 m in diameter. The sediment distribution maps (scale 1:10,000) are available at the GeoSea Portal of the BSH (www.geoseaportal.de) with different levels of detail (number of sediment classes). As an example, Fig. 4.3 shows the data set of Tauber and the hydroacoustic sediment mapping in comparison with the data set presently provided by the European Marine Observation and Data Network (1:1 M) (EMODnet 2019) for a selected area at the Western Rönnebank. The new high-resolution sediment maps based on backscatter data provide the opportunity to initiate, for example, more systematically benthic sampling and precise habitat modelling.

Surface sediments, especially the fine fraction (<63 μm), are reservoirs for chemical elements and nutrients. From geochemical analysis of the fine fraction on a selected subset of 800 grab samples within the German Baltic Sea, three functional element groups and their characteristic distribution patterns were identified (Leipe et al. 2017). The first group described by Leipe et al. (2017) are the biogenically relevant components representative for in the first order, primary and secondary production (TOC, N, P, biological opal). As expected, the total organic carbon (TOC) in the total sediment (all grain size classes) correlates with the mud content (Leipe et al. 2017). Surprisingly, if related to the fine fraction only, high TOC values are particularly high in sandy sediments with low mud content. This is the case in the Oder Lagoon and in the Pomeranian Bight (Fig. 4.4). Leipe et al. (2017) conclude that high TOC values are not necessarily related to high primary production, but conversely, high net primary production is always related to elevated TOC. The easternmost part of the Oder Bank shows, e.g., high TOC values but low primary production, suggesting that biogenic components are not autochthonous but were

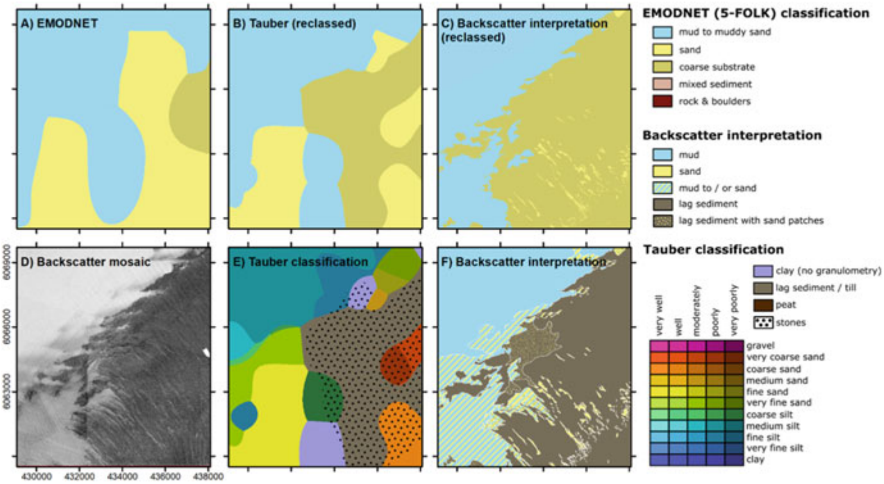

Fig. 4.3 Comparison of different sediment maps in the region of the Western Rönnebank. (**a**) EMODnet (2019) data set with five classes, (**b**) Tauber (2012a–i) data set reclassed after EMODnet (2019), (**c**) backscatter interpretation reclassed after EMODnet (2019), (**d**) backscatter mosaic (1 m resolution, dark colors: high backscatter), (**e**) original data set of Tauber (2012a–i), (**f**) original interpretation of backscatter mosaic after the mapping guideline of the BSH (2016)

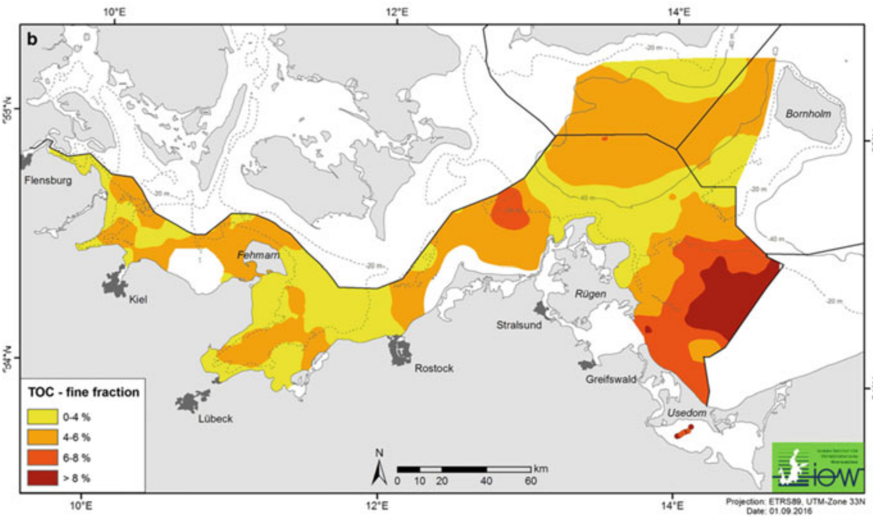

Fig. 4.4 Distribution of organic carbon (weight % TOC) in the sediment fine fraction (<63 μm) after Leipe et al. (2017)

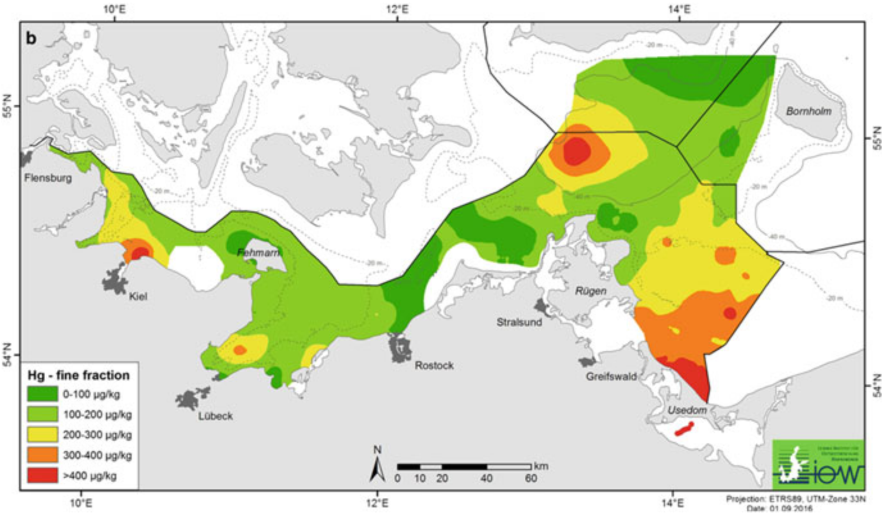

Fig. 4.5 Distribution of mercury (Hg, µg/kg) in the sediment fine fraction (<63 µm) after Leipe et al. (2017)

introduced by lateral transport from potentially more eutrophic sites (e.g., Oder River Mouth) (Leipe et al. 2017).

Similar patterns are described by Leipe et al. (2017) for the second functional group with at least partly redox-sensitive elements, being representative for eutrophication (Fe, Mn, P, S). They observed highest values in the sandy areas of the Oder Bank. In other sandy areas and in the muddy areas low P values are described. Leipe et al. (2017) consider that degradation of organic matter can be one reason for the distribution pattern and the missing correlation with mud content. P is mainly bound to iron oxyhydroxide and can be mobilized by early diagenetic iron reduction. The third functional group described by Leipe et al. (2017) comprises the anthropogenically sourced environmental pollution by heavy metals (Hg, Pb, Cu, Zn, and As), which can be used also as tracer to identify spot sources and dispersal pathways. Highest Hg concentrations within the fine fraction are restricted to confined areas, e.g., dumping sites or river outlets (Leipe et al. 2017). Examples are the dumping sites of industrial and post-World War II waste in the Mecklenburg Bight and the Arkona Basin, respectively (Fig. 4.5). Through sediment resuspension and lateral transport to the deeper part of the basins, the contaminant concentrations are generally decreasing with increasing distance to the sources (Fig. 4.5).

Hydro- and sediment dynamics lead not only to a lateral transport of sediment and elements but also cause a vertical mixing of sediments, which can blur the geochemical signals and potentially affect the chronological order of deposition. Bunke et al. (2019) have investigated the sedimentary impact of different natural and anthropogenic mixing processes (Fig. 4.6). Hydroturbation by storms or extraordinary bottom currents (e.g., in case of inflow events) potentially can affect the upper 8–10 cm of the sediment producing erosional surfaces and successions of partly graded cm-scale

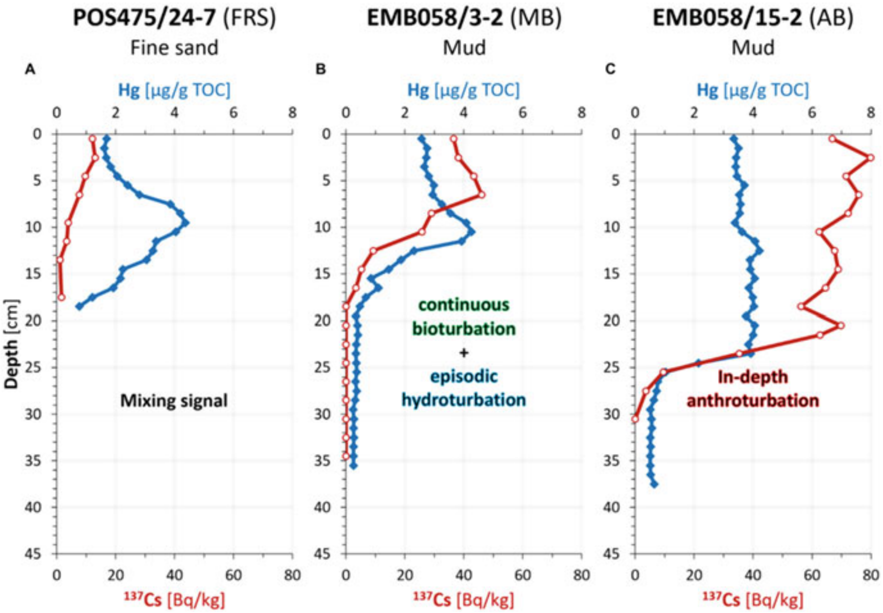

Fig. 4.6 Examples of how different sediment mixing modes do affect 137 Cs (red) and TOC normalized Hg (blue) profiles in short cores from (**a**) the Falster-Rügen sand plain (FRS), (**b**) the Mecklenburg Bight (MB), and (**c**) the Arcona Basin (AB) (after Bunke et al. 2019)

subparallel sediment layers. Subsequent repopulation and bioturbation by benthic organisms may partly or totally erase such structures in the upper 5–7 cm (Morys et al. 2016). Ichnofossils from mainly infaunal bivalves, crustaceans, and polychaetes can witness such bioturbation activity. Both hydro- and bioturbation processes tend to dampen the geochemical signal but may preserve the general stratigraphic order. This is very different for anthropogenic mixing like bottom trawling, which in many areas of the Mecklenburg Bay and the Arkona Basin has left behind a strongly furrowed sediment surface (Bunke et al. 2019). Homogenized sediments down to 25 cm and the lack of characteristic vertical element profiles are most likely indicative of more recent trawling impact.

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Tauber F (2012e) Meeresbodensedimente in der deutschen Ostsee: Kieler Bucht – Flensburger Förde, Karte Nr. 2931. Maßstab 1 : 100 000

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Environmental Conditions at the Coast: The Terrestrial Ecosystems

5

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Abstract

In this chapter, the characteristics of the terrestrial hinterland of the Baltic Sea coast in Schleswig-Holstein and Mecklenburg-Vorpommern are described from a physical viewpoint. Additional information on the basic social and economic structures can be found in Sect. 5.4. The descriptions encompass the main topographic features inside the Baltic Sea catchment area, its fundamental geological and geomorphological structures and the respective patterns of land use and land cover. The main climate conditions are described and the text touches on the soils, hydrological characteristics, the shapes of river systems, and a comparison of nature conservation areas.

5.1 Delineating the Study Area: Basic Characteristics

The "hinterland" of the coast is as important as the coastline (Chap. 6) and the marine (Chap. 3) characteristics. Therefore, the ecological conditions of the hinterland of the German Baltic Sea coast are briefly analyzed, mapped, and described in this chapter. Furthermore, the unfolded features will be helpful in comprehending the demands, flows, and potentials of ecosystem services (Chaps. 19–26). A

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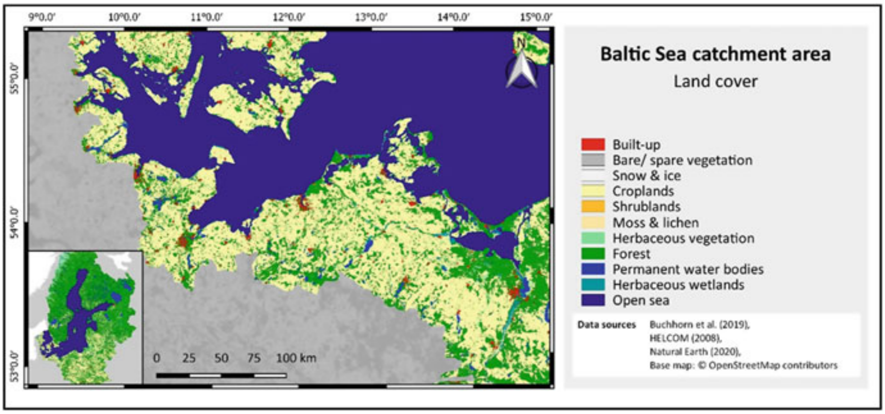

Fig. 5.1 The total watershed of the Baltic Sea (inset in the bottom left) and its German part with respect to general land cover structures. Additional sources: Swedish Meteorological and Meteorological Institute, HELCOM (2008), Buchhorn et al. (2019)

Table 5.1 Lengths of different coastal types and dykes in SH and MV

Schleswig-Holstein		Mecklenburg-Vorpommern	
Lengths of coastlines			
Length of coast total	536 km		1945 km
Main land coast Schlei Fehmarn	328 km 137 km 71 km	Outer coast Inner coast (Bodden coast) Darß-Zingster Bodden Rügensche Bodden Usedomer Bodden	377 km 1568 km 262 km 49 km 224 km
Steep banks, cliffs	122 km		351 km
Flat coast	414 km		1594 km
Lengths of dikes			
Dikes in total State dikes Regional dikes	121 km 69 km 52 km	Dikes in total Sea dikes Bodden dikes Protection dunes	218 km 45 km 173 km 106 km

Sources: Ministerium für Energiewende, Landwirtschaft, Umwelt, Natur und Digitalisierung des Landes Schleswig-Holstein (2014), Ministerium für Landwirtschaft, Umwelt und Verbraucherschutz Mecklenburg-Vorpommern (2009)

completion of the described site characteristics can be found in Chap. 13, where also some of the socio-economic features are described.

The German part of the hinterland in the south comprises only a small fraction of the catchment area of the Baltic Sea (40.012 km^2; Fig. 5.1) subdivided into the administrative watershed systems Schlei/Trave, Warnow/Peene, and Oder. Within the catchment, the Baltic Sea itself covers an area of 412,560 km^2. It is a shallow sea with an average depth of 52 m, while the deepest point has a depth of 459 m (for the Baltic Sea catchment in numbers see Table 5.1).

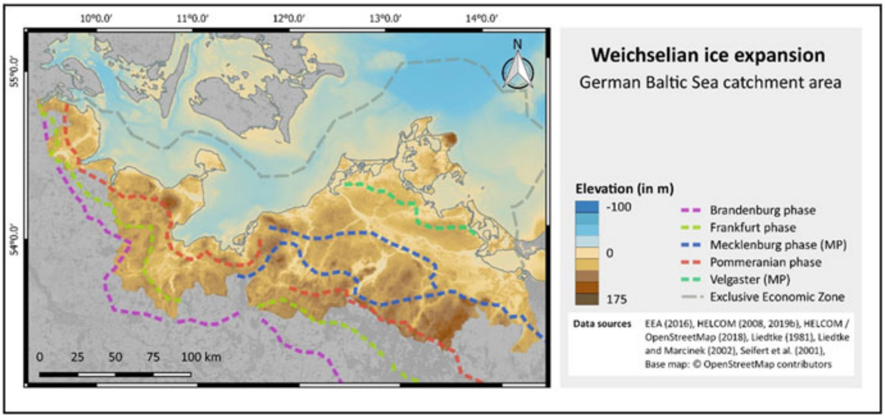

Fig. 5.2 Elevation and glacial edges of the last phases of the Weichselian glacial period; Sources: Ehlers (1994), Böse et al. (2018), Liedtke and Marcinek (2002)

Today's relief of this area, with relatively low altitudes within an undulated landscape, has been formed mainly during the Weichselian glaciation period (see, e.g., Niedermeyer et al. 2011). Therefore, the basic geomorphological elements of the terrestrial landscape are ground moraines, terminal moraines, glacial flood plains, sandurs (sand-dominated glacial outwash areas), outwash fans, glacial meltwater valleys and glacial lakes. Maximum elevations are reached at the Bungsberg (167.4 m) in Schleswig-Holstein and in the "Helpter Berge" in the "Uckermark" with 179.2 m in Mecklenburg-Vorpommern. Conspicuous elongated structures with low elevations are the river valleys: the Schlei Fjord, Schwentine and Trave and the "Oldenburger Graben" in SH, and Stepenitz, Warnow, Recknitz, Ücker, Randow and Peene in Mecklenburg-Vorpommern (Fig. 5.2).

Germany has 2.247 km of coastline at the Baltic Sea (see Table 5.1) of which 536 km belong to the state of Schleswig-Holstein (SH) whereas the larger part, i.e., 1.945 km, of the coastline are located in Mecklenburg-Vorpommern (MV). Especially in MV large parts of the coast belong to the so-called inner coast, featuring the shorelines of the Bodden water bodies (semi-enclosed passes and lagoons).

5.2 Geology and Geomorphology

The Weichselian glaciation (115.000–11.700 years BP) is an important driver of the current landscape (more details in Chaps. 6 and 7). During this period, Scandinavia was covered by an ice sheet and Northern Germany was located at the southern boundary of this glacial expanse. During the later phase of the Weichselian glaciation, this ice sheet oscillated in its extent and formed four distinct recognizable interstadial periods (see Fig. 5.2):

- Brandenburg phase (24.000–22.000 BP)
- Pommeranian phase (18.200–15.000 BP)
- Mecklenburg phase (15.000–13.000 BP)
- Velgaster and Usedom edge positions as parts of the Mecklenburg phase in the north of the country

The maximum ice expansions left their traces in today's landscape in the form of geomorphological landscape features of the glacial series (Ehlers 1994; Böse et al. 2018; Liedtke and Marcinek 2002), i.e., mainly ground and terminal moraines. Within the young moraine landscape, also remnants of accumulative ice-based processes can be found in drumlins (elongated hills formed by the streamlined movement of ice sheets), kames (irregularly shaped hills composed of sand, gravel, and till, accumulated in a depression on a retreating glacier) or eskers (long, narrow ridges deposited by streams flowing on, within, or beneath a stagnant glacier). During and after the retreat of the ice, the melting water dynamics of the retreating glaciers played an important role in modifying the landscape of northern Germany. On the one hand, we can still find relics of subglacial tunnel valleys and large meltwater streams and on the other, alluvial fans, the extensive sandurs follow the terminal moraines in southern and western directions. They today make up the landscape type of "Geest," with sandy and less fertile soils.

These glacial features determine the development of coastal types (cf. Chap. 6):

- *Förden coasts* (Fördenküsten) have been formed by glacial erosion and peripheral accumulation processes during the Pleistocene. They can be found in Schleswig-Holstein (e.g., Flensburger Förde, Schlei, Kiel Förde, Eckernförder Bucht, Trave Förde).
- *Graded shorelines* (Ausgleichsküsten) are based on directed erosion and accumulation processes, with erosion-, transportation-, and accumulation-zones, which are regulated by dominating currents, producing a smoothened coastline over time periods of centuries to millennia. We can find these coastal types, e.g., in the segment between Kühlungsborn and Warnemünde.
- *Bodden coasts* (Boddenküsten) are characterized by Pleistocene island cores which have been subjected to grading processes, therefore producing inner and outer coastlines. The resulting lagoon systems are called Bodden or Haff. Examples are the Darß-Zingst-Bodden chain, the Greifswalder Bodden and the Salzhaff.
- *Cliff coasts* (Kliffküsten) and flat coasts (Flachküsten): appear on glacial materials which have been accumulated with different elevations. Both coastal stretches with accumulation and erosion are interspersed (Fig. 5.3, for Schleswig-Holstein). However, the overall coastal length with erosion is larger than that with accumulation (182 km vs. 128 km for SH, LKN SH (2017). On the other hand, budget calculations for the period 1878–2010 revealed for SH an overall average land loss of 5.5 ha per year, compared with an average land gain by accumulation of 6.4 ha per year. Thus there has been an overall gain of about 400 ha in Schleswig-Holstein. Referring to the cliff coasts, about 2/3 are suffering from

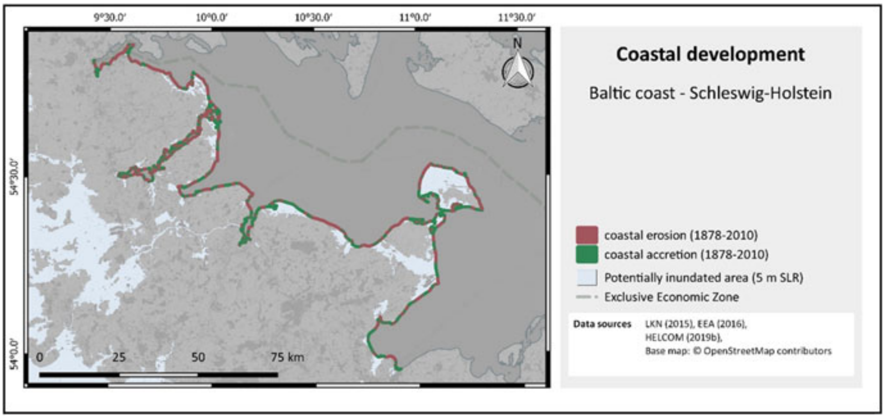

Fig. 5.3 Coastal erosion vs. accumulation in Schleswig-Holstein between 1878 and 2010; Source: Landesbetrieb für Küstenschutz, Nationalpark und Meeresschutz (2017)

retreat and abrasion. The flat coastal zones are endangered because 338 km^2 of flood-risk areas with ten thousands of inhabitants are situated behind the coastline.

5.3 Environmental Conditions

As a consequence of the geomorphological conditions, the actual land-use patterns show some typical flatland structures with a high proportion of agricultural activities. The dominant land use in both SH and MV is arable land (Table 5.2, Fig. 5.3), followed by pastures, forests and urban areas, but SH and MV differ considerably in the area of coniferous forest (MV > SH) and urban areas (Fig. 5.4). Cliff coasts cover a higher proportion in Schleswig-Holstein, whereas lagoons occupy a much bigger area in Mecklenburg-Vorpommern (see Table 5.2).

In Kiel and Rostock, average annual maximum temperatures are 11.9 and 11.6 °C with minima of 5.9 and 5.8 °C, respectively. Average annual precipitation amounts to 778 and 621 mm, respectively.[1] The average annual sunshine duration follows a strong gradient from north-east to south-west with highest values on the islands of Fehmarn, Rügen and Usedom (Fig. 5.5a). As a consequence of the sea surface temperatures, the elevations and the forest patterns, the sunshine duration is not directly related to the average annual air temperature distribution (Fig. 5.5b), for which there are maximum values in the region of Ostholstein, Fehmarn, Salzhaff, minima at the island of Rügen and in the south-eastern lake districts, while the water temperature forms a gradient from Flensburg Fjord to the Arkona Basin.

[1] https://www.dwd.de/DE/leistungen/klimadatendeutschland/klimadatendeutschland.html;

Table 5.2 Comparison of SH and MV with respect to land cover types. Note, that the resulting data are not typical for the whole federal states, but only for the study areas. The codes characterize the Corine land cover types, following the source GeoBasis-DE/BKG (2019)

Land cover code	CORINE land cover type	Schleswig-Holstein (%)	Mecklenburg-Vorpommern (%)
211	Non-irrigated arable land	34.24	31.82
231	Pastures	8.89	9.95
311	Broad-leaved forest	5.12	5.33
312	Coniferous forest	0.94	4.87
112	Discontinuous urban fabric	4.47	1.89
313	Mixed forest	0.41	0.90
835	Open coastal waters: seasonally stratified	25.70	26.83
833	Open coastal waters: non-vegetated sand, gravel, and sandbanks (type 1110)	3.19	6.43
622	Lagoons and Estuaries (types 1130 and 1150), WFD type B1 and B2: sandy, gravel, and sandbanks (type 1110)	0.24	2.50
623	Lagoons and Estuaries (types 1130 and 1150), WFD type B1/B2: non-vegetated clay and mud	0.41	1.85
831	Reef (type 1170)	9.62	1.53

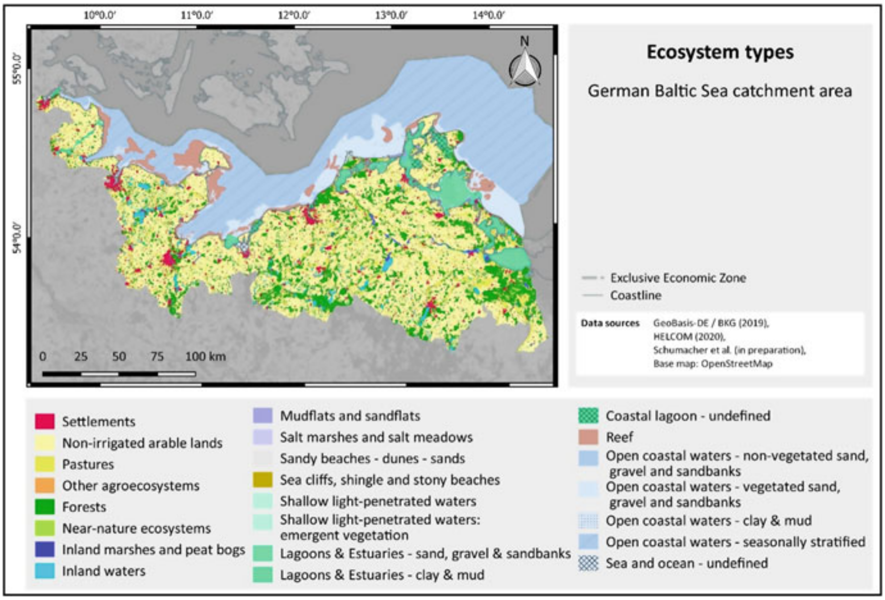

Fig. 5.4 Ecosystem and land cover classes of the marine and terrestrial study areas

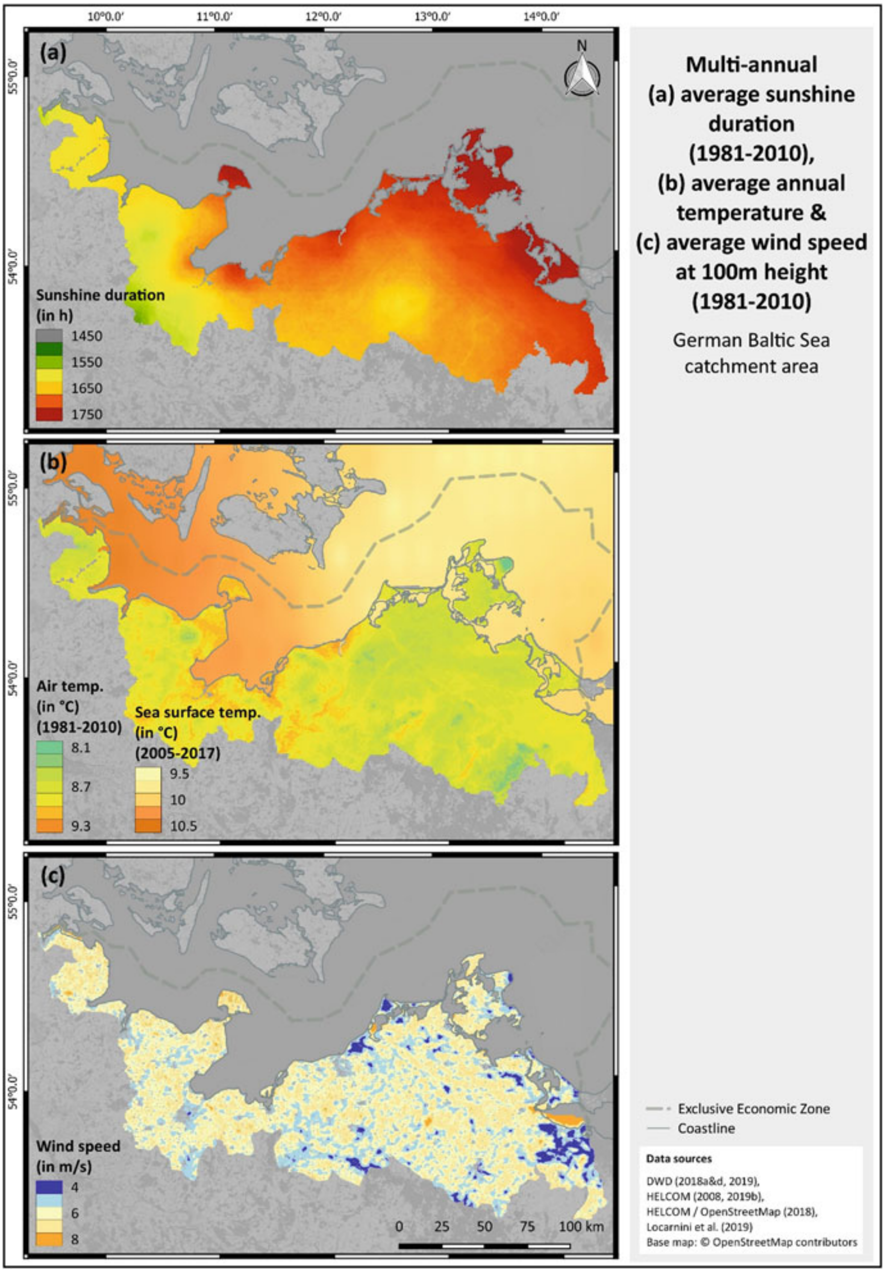

Fig. 5.5 Average sunshine duration, annual temperatures, and average wind speeds across the study area in the climate reference period 1981–2010

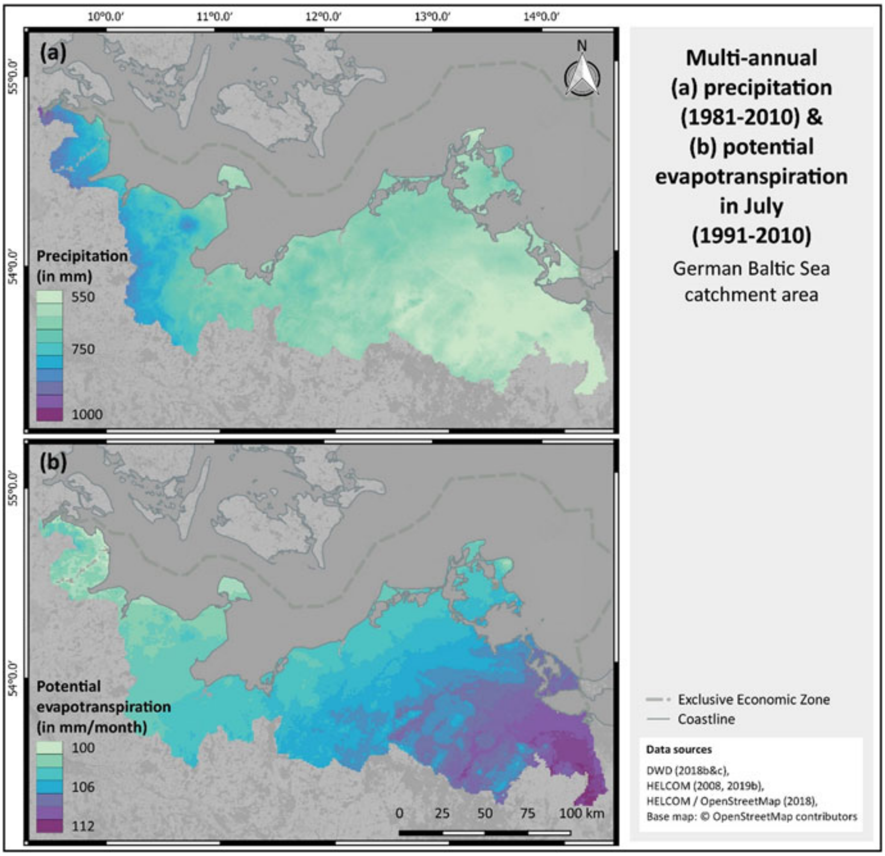

Fig. 5.6 (**a**) Average annual precipitation, and (**b**) average potential evapotranspiration in July for the climate reference period 1981–2010

The long-term average wind speed (Fig. 5.5c) shows in general high values in Schleswig-Holstein, local maxima, e.g., at Fischland or the Oder Lagoon and lower wind velocities in the Oder catchment area. Annual precipitation (Fig. 5.6a) shows a west–east gradient with average local values of more than 800 mm in Schleswig-Holstein while in the east, the precipitation falls below values of 550 mm per year. An opposite trend can be seen for the potential evapotranspiration in July (Fig. 5.6b). The highest values in the east are caused by high solar radiation input and high wind speeds. In contrast to potential evapotranspiration, the actual evapotranspiration will in most cases be much lower and follow Fig. 5.6b only in the cases of wetlands, including peatlands.

The potentials of ecosystems for nearly all functionalities and processes are strongly influenced by their soil conditions. In the marine part of the study area, soil textures (Fig. 5.7a) are dominated by muddy sediments in the SH area, while MVs coasts are dominated by sandy textures. On land sandy loam is the dominating

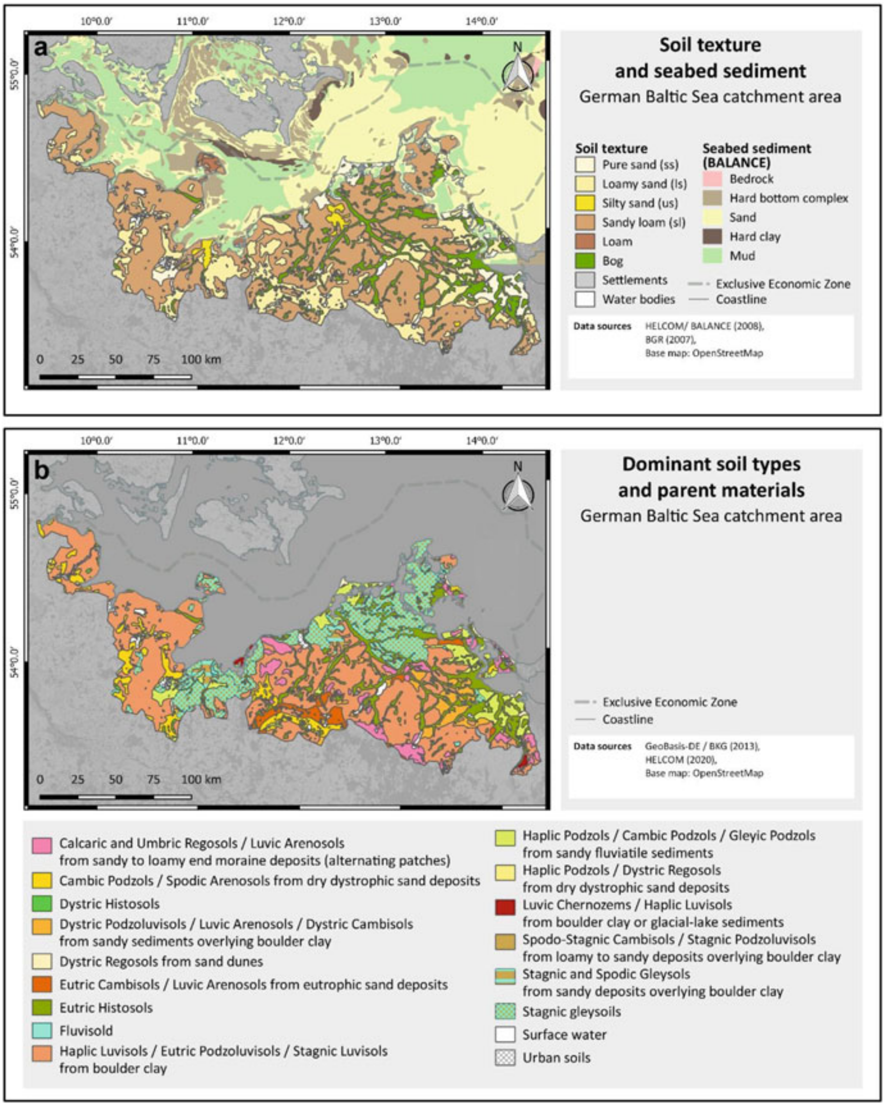

Fig. 5.7 (**a**) Soil textures and sediment classes and (**b**) soil types in the study area

texture, especially in areas, which developed from ground or terminal moraine material. The sandur areals and the inner fluvioglacial sediments are characterized by sand, whereas in the large old glacial valleys, mainly in Vorpommern, peatlands have formed during the Holocene.

Dominant soil types (Fig. 5.7b) are luvic arenosols and dystric cambisols on morainic materials, while on sandy material, haplic podzols, gleyic podzols and dystric regosol have formed. In the coastal vicinity of the eastern parts, large areas

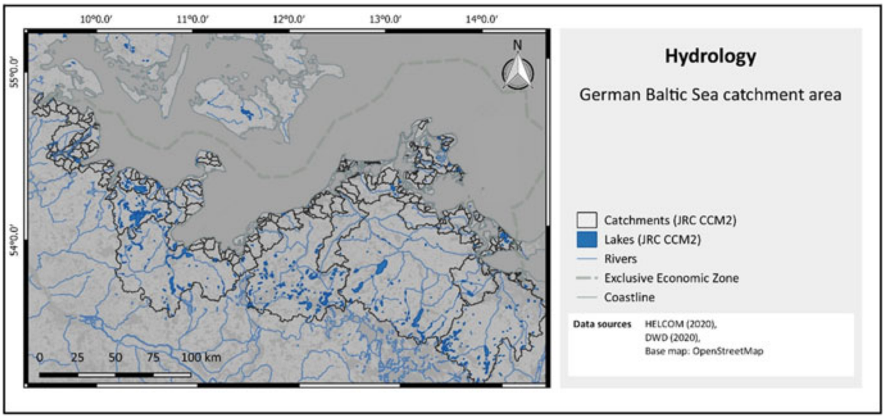

Fig. 5.8 Watersheds and river systems of the study area

show stagnic gleysoils, typical for undulated landscapes with relatively low elevation and high water dynamics. Most of the peatlands have been drained during the nineteenth and twentieth centuries leading to the formation of eutric histosols.

The river network (Fig. 5.8) is largely determined by the geomorphological history of the landscape. The map's shapes originate in the geomorphological history of the landscape resulting in watersheds of very different sizes. Glacial lakes appear in a regular pattern depending on the dynamics of the different ice expansions (Fig. 5.2). The characteristics of the main rivers are documented in Table 5.3. The Oder has the largest water input into the Baltic Sea, followed by Peene and Warnow. These are also the longest rivers. In contrast, the rivers in Schleswig-Holstein play a minor role.

The distribution of nature protection areas (Table 5.4) demonstrates that Schleswig-Holstein has a larger area of national parks than Mecklenburg-Vorpommern, mainly because of the Wadden Sea National Park at the North Sea, which is not relevant for our study area. On the other hand, the two federal states seem to follow different guidelines: while the eastern state provides a high number of nature reserves, protected landscapes, Natura 2000 areas and natural heritage areas, Schleswig-Holstein has a higher number of protected landscape areas and a slightly higher share of nature park areas.

With this information on the terrestrial parts of our case study region, it should be possible to assess and evaluate the forthcoming analyses and interpretations of the human-environmental systems and the ecosystem service distribution within the research area. In the following chapter, the dominating social and economic structures will be illuminated.

Table 5.3 Basic features of the administrative watershed areas (UBA 2016 [https://www.umweltbundesamt.de/steckbriefe-zu-den-flussgebietseinheiten (2016)])

	Size (km^2)	Population	Main rivers	Length of main rivers (km)	Average discharge (m^3/s) (1973–2012)	Dominant land use types (%)
Schlei/ Trave	9.218	1.25 Mio.	Füsinger Au, Kossau, Oldenburger Graben, Schwentine, Stepenitz, Wakenitz	Trave (113), Schwentine (70), Stepenitz (56)	- Trave: 8. - Schwentine: 4. - Stepenitz: 4.	Arable land (60), grassland (11), forest (11), wetlands (5), built area (5)
Warnow/ Peene	21.089	1 Mio.	Mildenitz, Nebel, Peene, Recknitz, Trebel, Warnow	Peene (125), Warnow (155)	- Peene: 22 (1961–2012). - Warnow: 17 (1989–2011).	Arable land (66), forest (18), built area (9)
Oder	9.705	0.68 Mio.	Bartsch, Lausitzer Neiße, Ohle, Stober, Warthe	Oder (162, total length: 855)	- Oder: 523 (1921–2013).	- Arable land: 48. - Grassland: 13. - Forest: 28. - Built area: 5.

Table 5.4 Characteristics of nature conservation activities within the study areas

Conservation area categories	Germany	Schleswig-Holstein	Mecklenburg-Vorpommern
National park areas (area in ha)	1,047,859	441,500	113,870
Biosphere reservations (area in ha)	1,994,276	443,100	113,700
Nature reserves (no.)	8676	195	286
Nature reserves (area share (%))	3.9	3.2	4.0
Protected landscape areas (no.)	8531	279	145
Protected landscape areas (area share (%))	27.9	14.8	30.3
Nature parks (no.)	103	6	7
Nature parks (area share (%))	27.9	16.4	14.6
Natura 2000 areas (area share (%))	15.4	10.1	29.2
National natural heritage areas (no.)	184	7	33

Source: BfN (2017)

5.4 Major Social and Economic Structures of the Research Area

The environmental state as well as the ecosystem service potentials and flows are strongly influenced by human activities in the two German Federal countries of Mecklenburg-Vorpommern and Schleswig-Holstein. As a starting point and in addition to Chap. 5, the distribution of the population has been compared in the map of Fig. 5.9. Here, besides the agglomerations of Flensburg, Kiel, Lübeck and Rostock, the big rural areas of Mecklenburg-Vorpommern become evident. Consequently, the population density of Schleswig-Holstein (183 n/km^2) surpasses Mecklenburg-Vorpommern (69 n/km^2) by two-thirds.

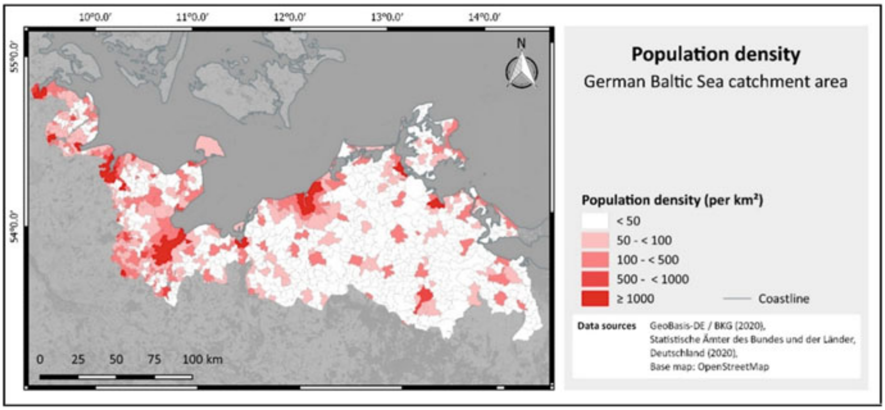

Fig. 5.9 The distribution of population density of the study area

The two federal states also differ strongly with respect to the economic structures. On the one hand, historical constraints and the assignments to different states during the cold war have caused these distinctions, on the other hand also the physio-geographic setting and the area sizes are responsible for some of the socio-economic gradients. These become obvious not only referring to the population and population density numbers, but also referring to GNP, employment rates and cargo handling figures (see Table 5.5). In both states, the economic branches with coastal backgrounds play important roles whereby Mecklenburg-Vorpommern has more employees in marine tourism, port management and marine technology, while Schleswig-Holstein predominates, e.g., referring to the significance of the navy and marine deliveries. Also the amount of handled cargo and the numbers of ferry passengers are higher in this state. Besides these distinctions both of the Baltic States reach a relatively small GNP compared with the average German values. Also the average incomes, the roles of manufacturing and industry are rather low, especially if compared to the dominant roles of services and the significant roles of tourism of Mecklenburg-Vorpommern and Schleswig-Holstein.

More than 42,000 (SH) resp. 34,000 (MV) persons in the study region are working in sectors which are highly related to marine ecosystems. Of course, one of the respective branches is shipping and marine traffic. With respect to the upcoming discussions on eutrophication or service demands, it may make sense to take a more precise look at the spatial distributions of these activities. Figure 5.10 shows the locations of shipping densities. In the smaller map, all ship movements have been registered. The Kiel Canal functions as a focal source of ships, mainly going to the north (directions Gothenburg, Oslo) or to the east, directing to the eastern Scandinavian and Baltic countries. It can also be seen that some marine traffic originates in Lübeck, Rostock and Szczecin. The bigger map shows the main abodes of the fishery fleets. They are much more widespread, and they provide additional centers in Heiligenhafen and on Rügen with special concentrations of fishing activities in the north of this island.

Besides the listed marine-economic activities, a central branch of employment in both countries of the case study area is tourism. Figure 5.11 illuminates the distribution of tourism, indicated by the numbers of touristic overnight stays. Both federal states had 30 Mio. overnight stays in 2017, whereby this number has been growing slowly in Schleswig-Holstein while Mecklenburg-Vorpommern das developed rapidly since the 1990s (around 10 Mio. Visitors in 1993). Obviously, the respective activities are scattered all over the coastal zone areas, but there are some focal areas, e.g., around the mouth of the Schlei fjord, on Fehmarn and around the Lübeck Bay, between Wismar and Rostock and on the island seascape of Darß-Zingst, Rügen and Usedom. Besides these areas, also the Lake District of Mecklenburg has many visitors, and in Schleswig-Holstein[2] also the North Sea coast attracts many people. Here the ranking of the top tourist municipalities in 2018 was as follows: Sylt

[2] https://www.statistik-nord.de/zahlen-fakten/handel-tourismus-dienstleistungen/tourismus/dokumentenansicht/tourismus-in-schleswig-holstein-im-januar-2020-61951/

Table 5.5 Some features of the economies in the study area

Economic features	Schleswig-Holstein	Mecklenburg-Vorpommern
Basic numbers		
Size of the area (km^2)	15.799.65	23.211.25
Inhabitants (n)	2.896.712	1.609.675
Population density (residents/km^2)	183	69
Unemployment rate (%)	4.9	6.7
GNP (Mrd. €)	93.37	44.91
Share of gross value added by manufacturing (%)	26	23
Share of gross value added by services (%)	73	76
Average annual income (1.000 EUR)	38	34
Economic growth (2017/2018, %)	1.8	0.7
Economic branches with coastal background		
Persons employed (n)	1.430.200	758.900
Employees navy (2013)	8.400	3.090
Employees marine deliveries (2013)	14.000	9.200
Employees shipbuilding (2012)	4.183	3.824
Employees fishery and fish processing	2.100	1.836
Employees marine tourism	3.000	4.000
Employees marine traffic and port management	5.800	9.359
Employees education and research	1.500	800
Employees offshore and marine technology	1.620	3.100
Seafaring numbers		
Cargo handling sea shipping (2013) (t)	35.856.000	25.645.000
Cargo handling Ro-Ro-ferries (2013) (t)	23.406.000	7.727.000
Ferry passengers (n) (2013)	14.031.678	2.997.712
Fish landing Baltic Sea (2017) (t)	11.630	15.633
Touristic numbers		
Overnight stays Baltic coast (2018)	13.683.423	24.850.608
Overnight stays Federal Country (2018)	30.251.579	30.884.000
Portion of German tourists (%)	15.8	19.8
Overnight stays per inhabitant (2018)	12	19
Touristic added value (Mrd. €)(2018)	4.5	4.1

Sources: Statistik Nord (https://www.statistik-nord.de/), Statistisches Amt Mecklenburg-Vorpommern (https://www.laiv-mv.de/Statistik/MV-in-Zahlen/), Sparkassenbarometer Touristik (https://www.tvsh.de/zahlen-daten-fakten/sparkassen-tourismusbarometer/), Kenngrößen der Volkswirtschaftlichen Gesamtrechnungen der Bundesländer 2018 (https://www.destatis.de/DE/Themen/Wirtschaft/Volkswirtschaftliche-Gesamtrechnungen-Inlandsprodukt/_inhalt.html), Statistische Ämter der Länder (2019), Johansen (2013)

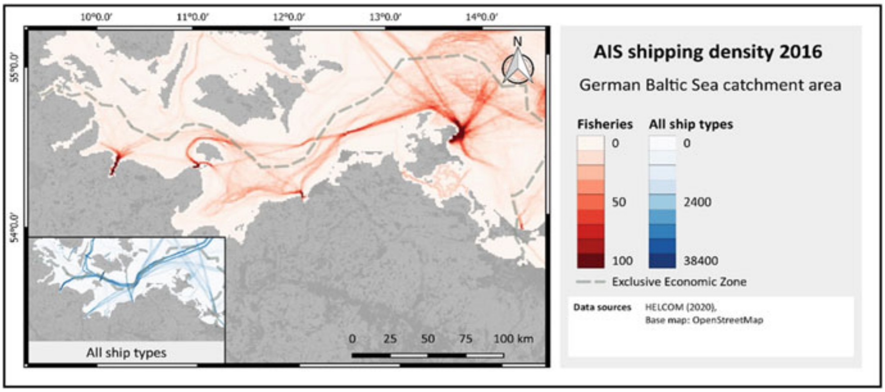

Fig. 5.10 Shipping densities of fishery boats (big map) and all ship types (small map) in the study region

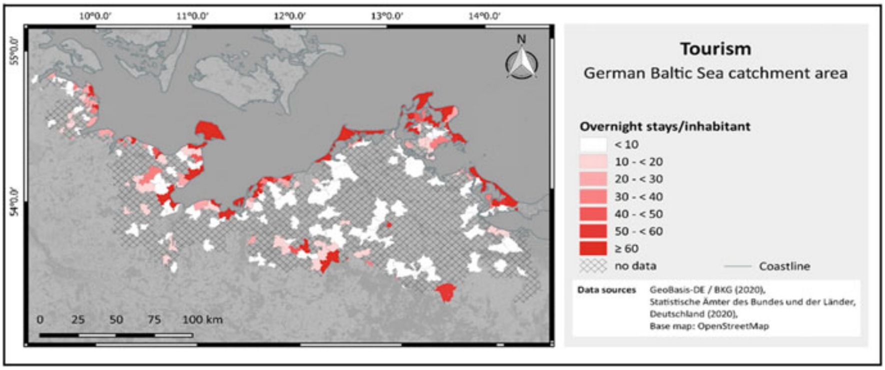

Fig. 5.11 Overnight stays in the municipalities of the study area

(2.9 Mio. overnight stays), Lübeck (1.8 Mio), Sankt Peter-Ording (1.5 Mio.), Grömitz (1.3 Mio.), Timmendorfer Strand (1.3 Mio.). Mecklenburg–Vorpommern[3] has developed towards becoming the most popular German holiday region since 2017, whereby the western Baltic Sea coast provides the biggest attraction with 8.2 Mio. guests in 2017 (27% of all overnight stays in Mecklenburg-Vorpommern), followed by Rügen/Hiddensee with 6.3 Mio. guests (21%), Usedom (5.3 Mio., 18%) and Fischland/Darß with 2.5 Mio. visitors (8%). The motivations of tourists to come to the German Baltic Sea coast are many and diverse (Table 5.6). However, many of the reasons listed are based on natural conditions and cultural ecosystem services, such as the marine recreation potential, the landscape character, the climate or the general quality of nature—as perceived by the tourists. Additionally, people come to

[3] https://www.tmv.de/uebernachtungen-nach-reiseregionen/

Table 5.6 Basic motivations of tourists to visit the study areas

Reasons for travelling to the baltic sea coast	Schleswig-Holstein (%)	Mecklenburg-Vorpommern (%)
Maritime aspects (beach, sea, bathing)	58	32
Recreation potentials	57	38
Landscape	52	41
Climate and air quality	47	66
Friendly inhabitants and hosts	35	9
Nature (flora, fauna)	30	41
Accommodation potential	30	7
Accessibility	28	19
Price	27	9
Cycling potential	26	21

Sources: Sparkassen- und Giroverband Schleswig-Holstein/Tourismusverband Schleswig-Holstein (2019) (https://www.tvsh.de/zahlen-daten-fakten/jahresberichte/), Ministerium für Wirtschaft, Arbeit und Gesundheit Mecklenburg-Vorpommern (2015, 2017)

these areas to find tranquillity, to enjoy the variability of the touristic offers, including arts, architecture, historical items, traditions, and cultural highlights.

Besides tourism, the land use by farmers of course plays an important economic role in the study area. Also from this viewpoint, there are several differences to be found in the agricultural structures of Schleswig-Holstein and Mecklenburg-Vorpommern: A focal example of these differences is the average farm size. While in Schleswig-Holstein, it lies at 77.9 ha, in Mecklenburg-Vorpommern this number is 274.9 (see Table 5.7). The reason can be easily related to the distinct political developments with private, smallholder related farms in the west while big agro-production communities have been established in the eastern part of Germany. These conditions in addition to the distinct sizes of the states' areas influence the numbers of farms. Further impressive differences occur with respect to the local innovations, e.g., by organics farming (4.7% vs. 17.2%), the average land price (27,100 € vs. 19,600 €), the number of livestock units (105 vs. 40) or meat production (193,600 t vs. 79,700 t).

The physical and human structures summarized above strongly affect the ecological conditions in the coastal zone area. The respective conditions are effectively determined by interrelations between the involved ecosystem types. So we can find several land-based influences in the sea, which come into existence due to flows of water, energy, and matter, due to inputs of nutrients, litter, pollutants, or deposits. In Fig. 5.12, some of these objects are depicted. Figure 5.12a demonstrates the German water discharge development flowing into the Baltic Sea from 1994 to 2016. Comparing this sketch with Fig. 5.12b and c makes clear that the overall freights of the nutrient elements N and P are extremely correlated with the water transfers. The Fig. 5.12d makes a distinction and concentrates on the N inputs from two important watersheds, the Schlei/Trave area in Schleswig-Holstein and the Warnow-Peene zone in Mecklenburg-Vorpommern. The first one provides lower

Table 5.7 Agricultural attributes of the study areas

	Germany	Schleswig-Holstein	Mecklenburg-Vorpommern
Number of farms in 2017 (no.)	266.690	12.460	4.900
Average farm size in 2017 (ha)	61	77.9	274.9
Proportion of organic farms in 2017 (%)	9.9	4.7	17.2
Area under agriculture in 2017 (Mill. ha)	16.66	0.99	1.35
Proportion of forest areas in 2017 (%)	34	11	24
Arable land (1.000 ha) in 2018	11.730	663.5	1.073
Permanent grassland (1.000 ha) in 2018	4.713	317.7	270.2
Average land price (€/ha) in 2017	22.300	27.100	19.600
Yield of cereals (dt/ha) in 2017	70.3	72.4	84.9
Yield of winter rape (dt/ha) in 2017	32.7	35.6	29.7
Yield of potatoes (dt/ha) in 2017	467.9	440.9	407.8
Yield of Silage maize (dt/ha) in 2017	474.6	423.6	413.9
Lifestock units per 100 ha LF in 2017	78	105	40
Cattle (tot., 1.000 animals/100 ha LF) in 2017	11.949	1.050	497
Pigs (1000 animals/100 ha LF) in 2017	26.445	1.414	833
Meat production (tot. 1.000 t) in 2017	6.650	193.6	79.7
Agriculture gross value added (Bil. €) in 2017	23.19	1.08	0.97
Labor force in agriculture in 2017 (no.)	940.100	39.800	23.900

Sources: BMVEL (2016, 2017): Daten und Fakten (https://www.bmel-statistik.de/) 2017, 2016, Statista (https://de.statista.com/themen/147/landwirtschaft/), Deutscher Bauernverband (https://www.bauernverband.de/faktenchecks), Statistikportal des Bundes und der Länder (https://www.statistikportal.de/de), Gehalt.de (https://www.gehalt.de/), Ministerium für Landwirtschaft und Umwelt Mecklenburg-Vorpommern (2019): Statistisches Datenblatt 2019

values in general, showing especially big differences in the early 1990 years. In both watershed areas, the share of diffuse sources has been declining over time.

Following UBA (2009)[4] and Behrendt et al. (2003), the nitrogen reaching coastal waters between 1998 and 2000, originated in groundwater by 55.7%, drainage outputs (15.4%), erosion and runoff with 3.9%, and from deposition processes with 2.2%. In general diffuse sources provide up to 80.9% of nitrogen to the coast, whereas point sources provide 19.1% (sewage plants 16.8%) and agriculture approximately 6.5% (own estimates based on available data).

[4] https://www.umweltbundesamt.de/daten/wasser/ostsee/flusseintraege-direkte-eintraege-in-die-ostsee#daten-zu-nahrstoffeintragen-aus-den-anliegerstaaten

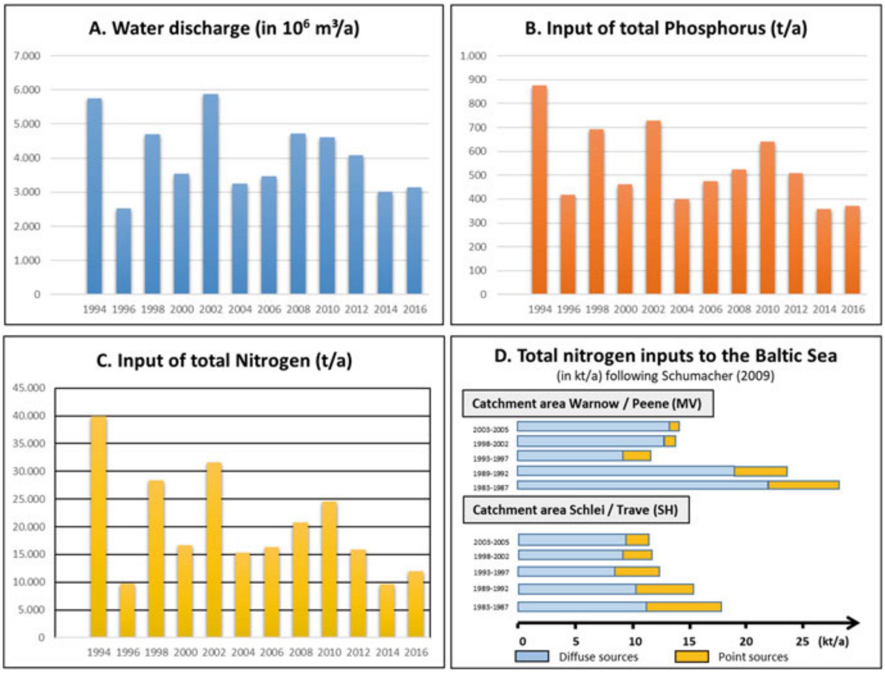

Fig. 5.12 Overall German water and matter inputs into the Baltic Sea; Sources: Schumacher (2009) and UBA web page (https://www.umweltbundesamt.de/daten/wasser/ostsee/flusseintraege-direkte-eintraege-in-die-ostsee#daten-zu-nahrstoffeintragen-aus-den-anliegerstaaten)

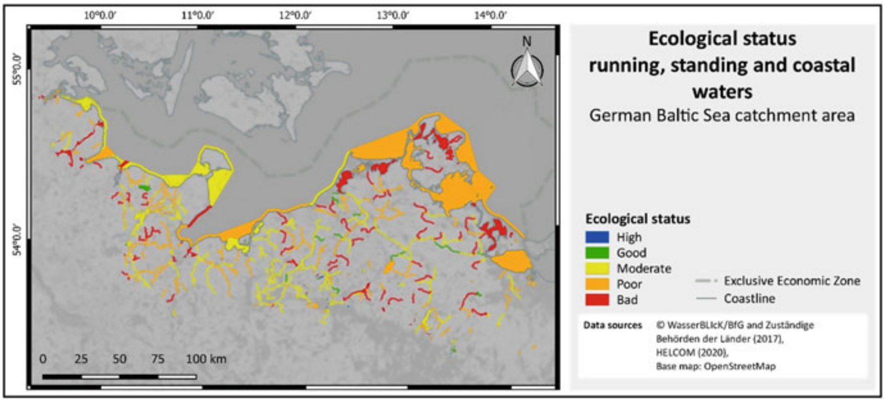

Fig. 5.13 Ecological status of coastal waters at the German Baltic Sea. Source: Bundesregierung (2018)

These input pathways are responsible for the overall status of the Northern-German waterbodies. Figure 5.13 shows that more or less none of the ecosystems jas high ecological status, a small group of rivers and lakes is in good status while the majority of the waterbodies falls into the categories moderate, poor, and bad. Especially the inner coastal ecosystems suffer from strong loads of nutrients and often are in bad shape. There is no coastal marine area which has a high or good status. This mostly eutrophication-driven situation is amplified by land-sea flows of heat, noise and information.

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6 Environmental Conditions at the Coast: Shoreline Ecosystems

Gerald Jurasinski and Uwe Buczko

Abstract

This part addresses the environmental conditions at the Southern Baltic Sea coast shoreline ecosystems providing knowledge of the natural conditions on which anthropogenic alterations act. The chapter starts off with some details on the geological formation of these coasts that build the geomorphological template on which ecosystems have established. Since this region has been covered by ice during the last glaciation, all landscapes and, thus, also the coastal ecosystems are comparably young. Due to the post-glacial dynamics, these coastal ecosystems are themselves highly dynamic on several temporal scales. Whereas cliffs change on millennial scales, dunes are much more dynamic and change on annual scales. In shallow low-lying sections peatlands developed and natural dynamics here would likely act on centennial scales. All the above temporal dynamics are strongly impeded by anthropogenic activities and can only rarely be observed today in the wild with strong implications for the development of the shoreline ecosystems in the future. On coastal peatlands long-term anthropo-zoogenic influence lead to the development of an alternative ecosystem. While under natural conditions plant species of brackish grasslands were confined to small areas below cliffs, grazing in reed belts over centuries caused the development of extensive areas of brackish grasslands featuring brackish specialist plant species. Today, the latter are rather rare because many of these areas have been diked for more intensive agricultural use in the past century.

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6.1 Introduction

Besides the marine features (Chap. 8) and the characteristics of the "Hinterland" (Chap. 5), a coastal systems analysis has to consider the coast itself, i.e., the zone that is influenced by both the "Hinterland" and the sea. The coastline forms a unique sea–land transition zone due to the exchange processes between environmental compartments (Jurasinski et al. 2018). The focus of the sea–land connection is often on extreme scenarios, for instance, flooding events (Jurasinski et al. 2018). Estuaries are another focus of coastal exchange research, whereas the coastline itself is rarely considered as an exchange interface for energy, water, and substances, yet this interface is very important as it operates continuously and may have far-reaching effects on (micro-)biological and hydro-biogeochemical processes on either side of the coast (e.g., Rullkötter 2009; Gätje and Reise 2012).

6.2 Southern Baltic Sea Coastal Geology

As a basis for understanding ecosystem and landscape structures some additional information to the basic geological background, given in Chap. 4, is necessary. The German coast of the southern Baltic Sea is part of the northern German lowlands, which has experienced a long-term trend of subsidence (since about 250 Ma, Permian, and still active today). Superimposed on this long-term trend are isostatic movements of the crust after the last ice age in combination with eustatic sea level rise. Since large parts of the German Baltic Sea coast are located south of a deep-reaching rupture zone which runs in NW-SE direction through Skane, the southern Baltic Sea north of Rügen, Koszalin, Bydgoszcz, Warszawa, and Lublin, the coast is slowly sinking with up to 1.6 mm per year (Lampe et al. 2010). In contrast, the major part of Scandinavia and the Baltic Sea proper shows a long-term uprising trend. The boundary between rising Scandinavia and the subsiding German lowlands is known as the Tornquist or Teisseyre-Tornquist Zone.

South of this zone stretches a thick sequence of Mesozoic and Cenozoic sediments. The surface sediments in MV and at the southern Baltic coast originate from the youngest (Vistula) glaciation, with very few exceptions, whereas north of it, the exposed surface rocks consist of old (paleozoic and often much older) magmatic and metamorphosed sequences (Pharaoh 1999). This has important ramifications for the coastal types in different parts of the Baltic Sea: Whereas in Scandinavia, the coast consists in general of hard rock formations with a well-differentiated coastline, the coast of the southern Baltic Sea is formed mainly by unconsolidated sediments which are prone to erosion and consequently result in a smooth, graded shoreline ("Ausgleichsküste," Lampe et al. 2007), and Fjord coast (Fördeküste) in SH (see also Chap. 3). This graded shoreline is of very recent origin and has developed only after the last glaciation, since about 10,000 years BP. Immediately after the Weichsel glaciation, the landscape of the present coastline consisted of an irregular, hilly topography formed by a succession of glacial moraines and troughs. Due to the eustatic sea level rise, marine waters entered the Baltic from the North Sea at

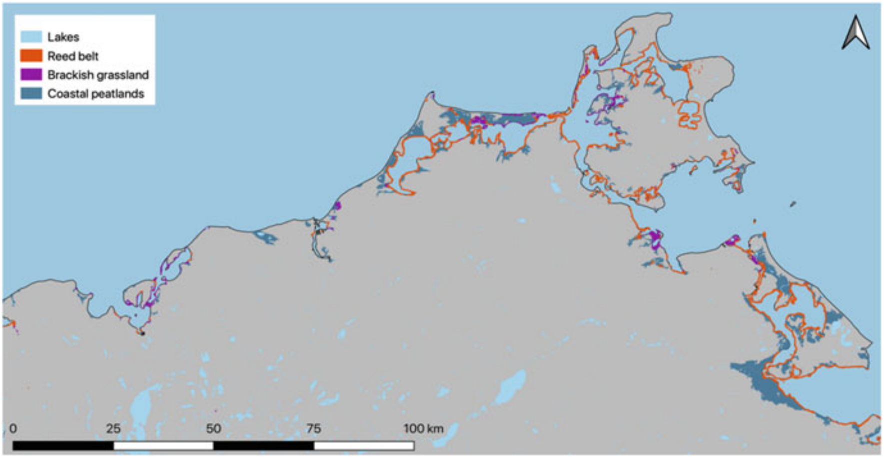

Fig. 6.1 Baltic Sea coast of Mecklenburg-Vorpommern with shallow low-lying areas dominated by reed belts, brackish grasslands, and coastal peatlands. The latter either formed as reed belt or as brackish grassland but was then cut off from the coast by a dyke. Map created by G. Jurasinski based on publicly available data sourced from https://www.umweltkarten.mv-regierung.de. Original data sources: Lakes—Technical information system for water bodies (DLM25W), LUNG MV 2015; Reed belt, Brackish grassland—Official map of biotopes and geotopes (BNTK), LUNG MV 2015; Coastal peatlands—Konzeptbodenkarte 1:25.000 (KBK25), LUNG MV 2016

8.000 BP via the Danish Straits (Great Belt, Little Belt and Oresund), and the brackish water Littorina Sea stage began. Only with the Littorina Transgression, the present graded coastline started to develop: Under the influence of west-east marine currents (induced by predominantly westerly winds), peninsulas formed and moraines were eroded, and interjacent bays were filled with sediments (Börner et al. 2019).

Today, the eroded parts form the cliff coast, at which the yearly coastal erosion rate may be as high as 50 cm even today. The eroded sediments are transported parallel to the coast and are deposited at coastal areas with weaker currents to form sandy beaches and sand dunes. At such coastal stretches, the sedimentation may result in a seaward progradation of the coastline of up to 1 m per year (at the beach of Warnemünde for instance). Further to the east (Darss), coastal sedimentation resulted in the graded Bodden coast of Vorpommern, with characteristic peninsulas (Nehrungen) and shallow brackish lagoons (Bodden, Haff).

The larger part of the total coastal length is comprised of “inner coast,” i.e., within estuaries and lagoon systems, which are prominent especially in MV. Only 377 km of the total costal length of MV is outer coast, whereas 1568 km are protected inner coasts (MLUV-MV 2010). These inner coasts are almost entirely lined with coastal wetlands of varying widths (Fig. 6.1), which are dominated by common reed (*Phragmites australis* (Cav.) Trin. ex Steud.), like other regions along the Baltic sea (Dijkema 1990; Karsten et al. 2003; Selig et al. 2007; Meriste et al. 2012; Altartouri et al. 2014). The area of coastal wetlands along the Baltic Sea is not precisely known. Sterr (2008) reports for the German part an estimated area of about 1800 km^2.

6.3 Cliffs

Prominent examples for cliff coasts are the Stoltera cliffs and the Nienhagen forest west of Rostock. Ecologically, cliffs are interesting as habitat for specialized avi- and arthropod fauna as well as for a few specially adapted plant species. Although most cliffs are protected by law, they are—under natural conditions—essentially ephemeral on a centennial to millennial basis, but the general public often views them as stable. In the literature the distinction between "active" and "passive" cliff developed with the first denoting cliffs that are in the direct influence of the sea waves, whereas passive cliffs are not directly influenced by the sea anymore, for instance, due to coastal uplift or decreasing sea levels. Passive cliffs at the coast, like the Königsstuhl on the island of Rügen, can still be reached and eroded by higher storm surges. Protecting cliffs from erosion is therefore a high priority of contemporary politics. The vegetation below the cliffs is comparable to the areas closest to the sea in the dune series (see next section), whereas the vegetation at the cliff and especially at the escarpment is typically not specific for marine environments but is driven by the respective bedrock at the cliff.

6.4 Dunes

Eolian sand dunes are frequently found along the outer coast. They are in general very young sediments of Holocene age. A prominent example is the conspicuous dune and strand sediment sequence of the Neudarss, which started to develop around 3000 years ago. Previously, during the Littorina transgression, a cliff had developed at the then northern edge of the Darss. This cliff became inactive around 3.000 years ago due to a change in the coastal sediment transport system concomitant with diminished sea-level rise rates (Schumacher 2000). Since then, about 120 well-preserved beach ridges accumulated in a cuspate foreland and the shoreline prograded in northern direction, forming the Neudarss. At the Darsser Ort, the northernmost tip of the Darss peninsula (Fig. 6.1), the recent natural dynamics of dune series formation (see Fig. 6.2) can be studied, since the area is protected as core zone of the National Park "Vorpommern Bodden coast." At the "Weststrand," some of the former dune valleys are now developing as coastal paludification fens (see below for details on this type of coastal wetland).

The vegetation on sand dunes strongly changes within small distances from the coast (or better from the mean annual water level) following a distinct sequence of plant community types (Fig. 6.2). While the wash margin features ephemeral appearances of coastal vegetation, different dune vegetation types develop driven by the horizontal and vertical distance from the shore line. These are typically displayed along a regular transect but in reality the arrangement of the different

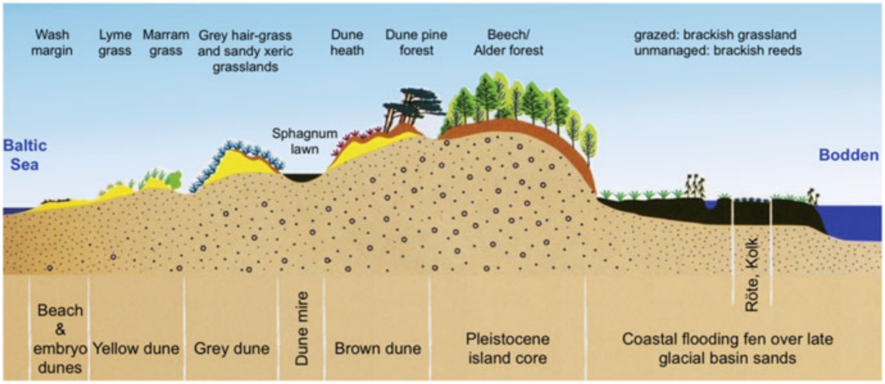

Fig. 6.2 Typical succession series of dune vegetation at the Baltic Sea coast of Mecklenburg-Vorpommern. Source: Umweltministerium Mecklenburg-Vorpommern (2003)

types is much more patterned in space as can be observed very well in one of the last natural dune-forming areas on the Northwestern tip of the Darss peninsula.

Due to its graded nature, the Bodden coast features an outer as well as an inner coast with a typical sequence of vegetation types. Brackish grasslands or reeds very rarely develop at the outer coast. If so, they are still protected behind small natural dunes and are only episodically flooded with sea water when storm surges breach through the dunes. As a consequence, coastal paludification fens develop. In contrast, coastal flooding fens developed at the majority of the shores of the inner coasts.

6.5 Shallow Low-Lying Coast

Large parts of the German Baltic Sea coast are graded and shallow. These areas are characterized by a relatively wide ecocline from land to sea (Jurasinski et al. 2018) and are, under natural conditions, often covered by extensive wetlands with specific flora and fauna. Natural coastal wetlands deliver many ecosystem services including coastal protection by providing space for water retention during storm surges, carbon storage and sequestration, retention and conversion of sea-borne nutrients, faunal and floral biodiversity (e.g., Narayan et al. 2017). In their assumed natural state, coastal wetlands of the German Baltic Sea Coast are thought to be dominated mainly by Common reed (*Phragmites australis*) and other emergent macrophytes tolerant to brackish conditions, like *Schoenoplectus tabernaemontani* (Grey Clubrush) or *Bolboschoenus maritimus* (Sea Clubrush). From the early 1950s on, however, many of the coastal wetlands were cut off from the sea or from the Bodden by dykes and subsequently drained for intensive agriculture. Today coastal areas in the region are therefore either characterized by fringing reed belts, coastal flooding peatland, pasture, or meadow. Which vegetation develops in a non-dyked coastal wetland is mainly driven by the prevailing soil substrate, the salinity, the elevation and, thus, flooding dynamics, and by land use.

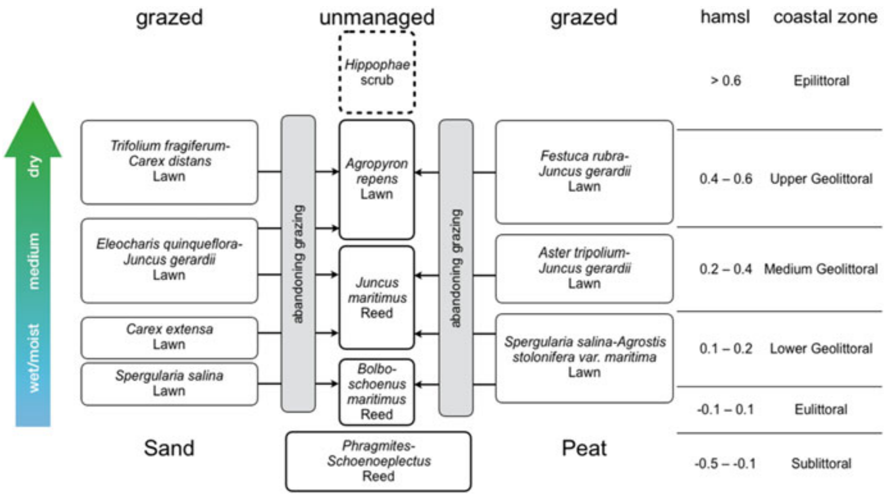

Fig. 6.3 Model of the vegetation development in the beta-mesohaline area of the Baltic Sea, as a function of substrate, position relative to mean water level and grazing (developed based on a Geolittoral only model from Jeschke 1987). There are slightly different vegetation types developing in the alpha-mesohaline (higher salinity) and the oligohaline (lower salinity) regions

Substrates dominated by sand cause a fundamentally different vegetation development on shallow, low-lying coasts compared to those dominated by silt (including organic sediments and peat, i.e., sedentary organic substrates) (Jeschke 1987). Upon abandonment, the vegetation communities converge and the differences between these two types of substrates disappear (Jeschke 1987, see also Fig. 6.3). Salinity is an important factor determining the distribution of halophytes (Dijkema 1990), because high salt contents require plants to develop effective adaptation mechanisms. The salinity of the Baltic Sea water along the German Baltic Sea coast decreases significantly from west to east, which leads to a large-scale zonation of the vegetation communities (Krisch 1974; Jeschke 1987). The colonization of the flat coasts with plants and animals is mainly determined by distance and exposure to the mean high tide line, as well as by the flooding frequency and the salinity of the soil (Seiberling 2003; Seiberling and Stock 2009). The lower the elevation of the flooded areas, the higher the influence of saline water. Therefore, elevation is an indicator for dominant plant communities (Jeschke 1987; Jutila 2001) (Fig. 6.3).

Since the Baltic Sea is almost free of a regular tidal range (Dijkema 1990), the delimitation of coastal zones is more difficult compared to the North Sea (Fig. 6.3). However, there are seasonal and inter-annual fluctuations in water levels, which have a significant influence on the establishment and development of vegetation in the Geolittoral (between 10 and 70 cm > MW). In the area of the Baltic Proper, which also includes the coastal sections considered here, there are only very small variations in mean water levels between spring and summer (Tyler 1969). However, there are marked variations between years, so that relatively low and high water level years alternate (Dijkema 1990). Low spring water levels are important events for the re-establishment of, e.g., *Schoenoplectus tabernaemontani* (Grey Clubrush) or *Eleocharis palustris/uniglumis* (Marsh or Slender Spike-rush) on areas not previously colonized (Dijkema 1990) but also low water levels in late summer and autumn in otherwise flooded coastal peatlands seem to trigger major vegetation re-establishment events (Koebsch et al. 2020). However, high water levels (MHW at 1.10 NHN) occur mostly due to floods in autumn and spring, as they are linked to storm events. Storm-induced high water levels naturally fall relatively quickly. Therefore, the effective discharge of water from any flooded coastal wetland area is of importance. Just as important as the elevation zonation and, thus, the access of sea water to a coastal wetland is the speed at which the water runs off again after flooding events.

Previous land use plays a central role in the release of nitrogen and phosphorus after rewetting and in the status of soil organic matter. These factors in turn not only influence biogeochemical conversions after rewetting, but also the establishment of plant species. A second dimension of land use lies in the fact that some of the vegetation communities typical of coastal floodplains (e.g., brackish grasslands) are suggested to be only able to develop in the presence of appropriate disturbance or grazing regimes (Jeschke 1987; Berg et al. 2004). It has been suggested, that most of today's salt—or rather ‚brackish—grassland at the Baltic Sea coast has developed as salt pasture through the influence of human livestock (Krisch 1974; Jeschke 1987; Berg et al. 2004). Jeschke (1987) estimates that only 10% of the area of approx. 13,000 ha previously populated by this vegetation type had been preserved as early as in the 1980s.

It is assumed that without any influence from land use, large parts of the shallow Baltic Sea coast would be characterized by reed belts instead, which colonized the coastal flooding fens at the bottoms of the coastal dunes and the shores of the beach lakes (Härdtle 1984). As early as in the thirteenth century, the coastal inhabitants began to use these areas for pasture, mainly with cattle. The treading of the cattle compacts the soil and promotes the incorporation of fresh litter, which is thus removed from direct decomposition (Jeschke 1987). This stimulates peat formation and allows the brackish grasslands of the southern Baltic Sea coast to grow above the mean high water line by means of peat growth, thus improving the site conditions for the development of salt grassland in a self-reinforcing process (Dijkema 1990). Therefore, it has been suggested that the brackish grasslands of the southern Baltic Sea coast can only be preserved in the long term through appropriate grazing (Jeschke 1987; Dijkema 1990).

Investigations after rewetting the polder Ziesetal (Seiberling and Stock 2009) show that grazing of too low intensity (less than one livestock unit per ha) can lead to monotonous *Agrostis stolonifera* floating lawns. However, decreasing or ineffective grazing with less robust breeds alone can also lead to a rapid expansion of the reed belts, which in turn leads to a positive feedback through lower grazing pressure because the animals avoid areas with high reed growth (Sweers et al. 2013). Seiberling and Stock (2009) conclude that a grazing pressure of 1–1.5 livestock units per ha should be guaranteed if the development goal is species-rich brackish grassland. In addition, Sweers et al. (2013) conclude that grazing with water buffaloes seems to be suitable to push back reed belts and to achieve species-rich salt grassland. In addition, an early mowing of the previous year's overgrowth seems to have a positive effect, as it can further enhance the species spectrum (Seiberling and Stock 2009).

6.6 Coastal Reed Belts

Before the influence of humans took hold in these ecosystems many centuries ago, large areas of coastal wetlands were occupied by relatively uniform stands of large emergent macrophytes like Common reed, Sea Clubrush, Grey Clubrush and other similar species (Jeschke 1987). To date the reason behind the large-scale dieback of reed belts in the 80s and 90s of the last century is still not completely understood (e.g., Gigante et al. 2013). While coastal reed belts may be less interesting in terms of plant species diversity today, they have likely hosted more species than today in their natural form because of generally much lower nutrient loads. It is to assume that due to natural disturbance events species which today have their focal habitats in brackish grasslands occurred also in natural shallow coastal areas in an essentially ephemeral way following disturbances and the thereby instigated favorable conditions. Land-ward natural or near-natural coastal reed stands are today either replaced by Brackish grasslands or by pastures and meadows behind dykes, although they may have been biodiversity hotspots, not necessarily for plant species but for biodiversity in general, hosting a variety of highly specialized species, like the ground beetle *Agonum monachum monachum* and their complex communities (Schmidt and Trautner 2016) and can be found only very rarely.

6.7 Pastures and Meadows Behind Dykes

Early on, however, the large majority of low-lying coastal wetlands have been converted into agricultural land, mainly for grass and fodder production. Later, often starting in the second half of the nineteenth century, but also much later in the second half of the twentieth century, the large majority of these areas were diked and then drained and used as intensive grassland mainly for fodder production, leaving only small remnants of the anthropo-zoogenic brackish grasslands of the Baltic sea coast. As a result, wetland sediments behind the dikes—often organic peat soils or mineral sediments with high organic matter content—degraded because of a mineralization of the organic compounds. Along with the degradation of these organic soils goes a massive CO_2 release (Jurasinski et al. 2016), a modified nutrient cycling and a shift in the hydraulic properties (Liu and Lennartz 2019). In addition, agricultural treatment loaded the wetland soils with nutrients, mainly nitrogen and phosphorus. Often, the surface elevation of artificially drained coastal wetlands is now below mean high-water level because of the subsidence of peat soils during degradation.

In general, the productivity of coastal wetlands is low in an agricultural sense (not including freshly reclaimed tidal marsh soils) because of a difficult soil moisture management. Productivity decreases over time and soil management becomes challenging because of soil degradation, subsidence, and rising water levels. A continuous agricultural usage is bound to a constant maintenance of the artificial drainage system consisting of the tile-drain and/or ditch network as well as pumping stations. A shift in the perception of agricultural activity especially on marginal land such as coastal wetlands has led to the abandonment of agricultural fields in selected coastal regions, for instance, on the southern Baltic Sea coast. The decommissioning of pumping stations may result in elevated water tables (fresh water) and positive hydraulic heads from land to sea with possible sub-marine groundwater discharge (Jurasinski et al. 2018).

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Ecosystem and Landscape Functions of the Coast: Recent Research Results

7

Uwe Buczko, Svenja Karstens, Franziska Schwark, Claudia Tonn, and Gerald Jurasinski

Abstract

This part provides an overview of ecosystem and landscape functions of the Baltic coast based on recent research in the reed belt of the Darss-Zingst Bodden Chain, a sheltered lagoon system of the southern Baltic Sea. The coastline of these lagoons is dominated by common reed (*Phragmites australis*). Important physical and chemical functions of these shallow coastal ecosystems are: (1) erosion protection and vertical accretion, (2) carbon storage and sequestration, and (3) buffering of nutrients, especially phosphorus (encompassing the components sedimentation, sorption, precipitation, and plant uptake). Phragmites wetlands are very effective for erosion protection due to their dense rhizome network. Moreover, they can increase ground level elevation by biomass accumulation and sediment accretion. In the DZBC, the capacity to accrete sediments and biomass depends on the topography and land use of the hinterland. Carbon storage and sequestration are related to this vertical accretion. Sediment carbon stocks (down to 1 m depth) range between 8.3 and 37.7 kg C m^{-2}. Phosphorus dynamics in the reed belts is governed by sorption, sedimentation, and plant uptake. Whereas sorption of P is reversible and governed by short-term meteorological and hydrodynamic processes, P accumulation by sedimentation and plant uptake is regulated on a longer term time scale.

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7.1 Introduction

Coastal wetlands, especially at the inner coasts, can protect the land from erosion and may produce vertical accretion, but they can also function as nutrient buffers and carbon stores. The Darss-Zingst Bodden Chain (DZBC) at the German Baltic Sea coast is an example of a well-studied lagoon system (see Chaps. 9–18; Karsten et al. 2003; Selig et al. 2007; Lampe et al. 2010), which we will use in the following as an example for ecosystem and landscape functions. The DZBC consists of four sub-basins with a total area of about 200 km^2, but water depths are shallow with a mean depth of only 2 m, and maximum depth of 14 m (Schlungbaum 1982a, b). Since the only connection to the Baltic Sea is a narrow outlet in the northeast, and the main freshwater inputs are the rivers Recknitz and Barthe in the western part, it displays a west-east gradient with very different salinities, ranging from 0 to 3 PSU in the innermost (western) lagoon (Saaler Bodden) to 7–10 PSU in the outermost (Grabow) (Selig et al. 2007).

The total coastal length of the DZBC is 267 km (MLUV-MV 2010), formed by an almost continuous belt of *Phragmites* wetlands (Fig. 10.1), with an estimated area of 13.5 km^2 (1350 ha). Such reed belts are typical for the inner coasts of the southern Baltic Sea. Therefore, the DZBC can be considered representative for Phosphorus storage in brackish lagoons of the Baltic Sea. The DZBC has been reasonably well studied not only during past decades (Schlungbaum 1982a, b; Schlungbaum et al. 1994; Karsten et al. 2003; Schumann et al. 2006; Selig et al. 2007) but also more recently (Karstens et al. 2015, 2016a; Berthold et al. 2018). The recent studies focused on two sites, Dabitz and Michaelsdorf, which differ with respect to salinity, the width of the *Phragmites* wetlands, topography, and land use in the hinterland. Whereas at Michaelsdorf, the salinity is about 3 PSU, it is 7 PSU at Dabitz. The width of the reed belt is about 20–70 m at Michaelsdorf and 80–150 m at Dabitz. At Dabitz, the hinterland has been used as arable land at least since the 1950s, the topography is undulating with elevations of up to 20 m, and the arable fields are not confined by a dyke from the coastal wetland. In contrast, the hinterland at Michaelsdorf is flat, used as grassland for sheep pasture, and a dyke was built in the 1970s between the grassland and the coastal wetland (Karstens et al. 2016a). The reed belt at Dabitz shows a characteristic zonation of interior, basin, and fringe zone (Fig. 7.1), which is developed similarly at many locations within the DZBC.

7.2 Physical and Chemical Functions of Shallow Coast Ecosystems

Important physical and chemical functions of shallow coastal ecosystems which are discussed in detail here are: (1) erosion protection and vertical accretion, (2) carbon storage and sequestration, and (3) buffering of nutrients (encompassing the components sedimentation, sorption, precipitation, and plant uptake).

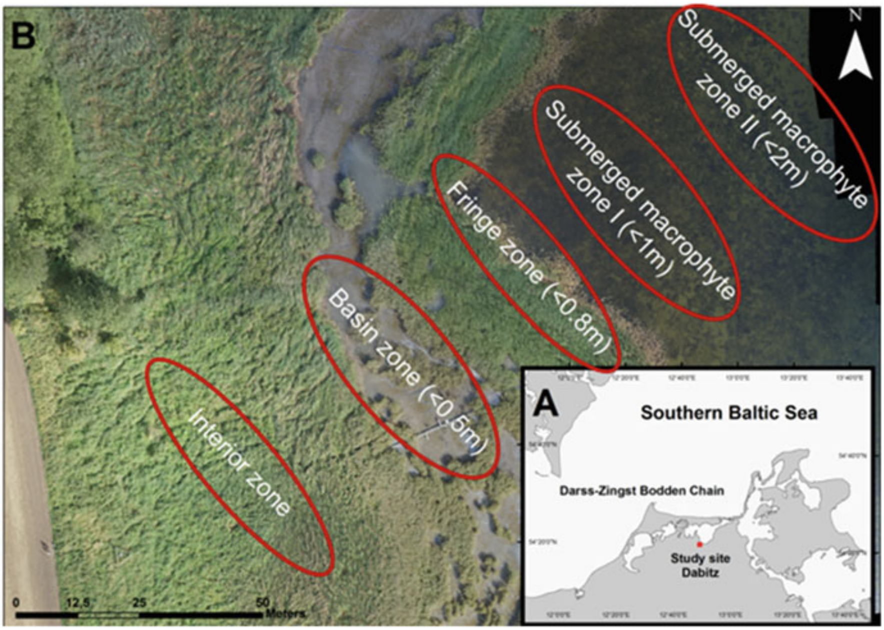

Fig. 7.1 Aerial view of the zonation within the reed belt at Dabitz (water depths in parentheses) (Berthold et al. 2018)

7.2.1 Erosion Protection and Vertical Accretion

Coastal wetlands can adjust to sea level rise (SLR) rates by vertical accretion of up to 12 mm y^{-1} (Morris et al. 2002; FitzGerald et al. 2008), depending strongly on biomass production and sediment particle delivery from land (Kirwan and Megonigal 2013). In the Baltic Sea Region, recent average SLR is 1.2 mm y^{-1} (Novotny 2007). However, this rate is spatially highly variable due to regional differences in isostatic movements and subsidence. In the Darß-Zingst Bodden chain (DZBC), average SLR during the twentieth century was only about 0.7 mm y^{-1} (derived directly from water level gauge measurements; Dietrich and Liebsch 2000). However, during the last 20 years (1993–2019), SLR in the Bodden region increased markedly to about 3 mm y^{-1} (calculated from satellite data; EEA 2019; https://www.eea.europa.eu/data-and-maps/figures/trend-in-absolute-sea-level).

In general, Common reed (*Phragmites australis*) has a high biomass production and can very effectively trap sediment particles—however, the ability to cope with rising sea levels strongly depends on the delivery of sediment particles from the hinterland. In the DZBC, several lines of evidence (i.e., sedimentation rates measured with the ^{137}Cs method, analysis of historical aerial images, fine-scale measurements of recent topography changes with SET) indicate that reed belt wetlands are at some locations (for instance, at Dabitz) able to keep pace with the recent SLR rates of 2.5–3 mm y^{-1}, and even prograding seawards, whereas at other locations (e.g., at Michaelsdorf), the Phragmites wetland cannot accommodate those SLR rates, and the coastline has been receding during the past decades. These differences seem to depend on the topography and land use of the hinterland, and the resulting differences in the delivery of sediment particles.

At Dabitz, a combination of high sediment input from the hinterland and favorable conditions of vegetation growth within the wetland allow the reed belt to accrete vertically and thus keep pace with SLR. In contrast, surface elevation changes at Michaelsdorf cannot keep pace with the current rates of sea level rise (Karstens et al. 2016b) and the coastline has been retreating during the past decades. The Phragmites wetland at this coastal stretch may be inundated in the coming decades with accelerated sea level rise and higher water levels have been associated with reed die-back, especially when the reed belts are monodominant (e.g., Gigante et al. 2013; Lastrucci et al. 2017). The main reason for the wetland retreat seems to be low sediment input from the hinterland, caused by the low elevation, the grassland cover, and the dyking since the 1970s (Karstens et al. 2016a). Since at Michaelsdorf, the Phragmites wetland is unable to migrate landwards due to the dyke, the width of the belt might diminish, and a "coastal squeeze" situation with a gradual annihilation of the wetland at this site could be expected (cf., Doody 2004; FitzGerald et al. 2008; Kirwan and Megonigal 2013).

7.2.2 Carbon Storage and Sequestration

Coastal wetlands play a particularly important role in sequestering and storing C, which is referred to as "blue carbon" (Nellemann et al. 2009; Crooks et al. 2011; McLeod et al. 2011; Pendleton et al. 2012; Luisetti et al. 2013). Although globally the total area of coastal wetlands is small compared with other ecosystem types, and consequently, the total amount of C stored globally in coastal wetlands is relatively small (1500 Pg in soils worldwide vs. 3–7 Pg in coastal wetland sediments, McLeod et al. 2011; Pendleton et al. 2012), the C sequestration rates are very high compared with other ecosystem types (on average 200 g C m^{-2} y^{-1} in coastal wetlands vs. 20 and 5 in inland wetland and forest ecosystems, respectively, Ouyang and Lee 2014). These high C sequestration rates are due to high biological productivity and the low rates of decomposition in waterlogged wetland substrates, accompanied by sea level rise and subsidence in many coastal wetlands (McLeod et al. 2011; Ouyang and Lee 2014).

However, coastal wetlands are endangered by human activities, such as urbanization, construction of roads, dykes and dams, aquaculture, sea level rise, and excessive nutrient input resulting in eutrophication (Crooks et al. 2011; Deegan et al. 2012; Hopkinson et al. 2012; Kirwan and Megonigal 2013). When coastal wetlands are converted into agriculture, aquaculture, or industrial use, the C stored in the sediments may be released back into the atmosphere, exacerbating the rise of atmospheric CO_2 concentrations (Pendleton et al. 2012; Bu et al. 2015). Although C storage in different types of coastal wetlands has been extensively studied (e.g., Chmura et al. 2003; McLeod et al. 2011; Pendleton et al. 2012; Ouyang and Lee

2014; Kulawardhana et al. 2015), large uncertainties still exist, due to the large variety of environmental parameters (e.g., salinity, nutrient status, sediment supply, climate, species composition, tidal range) that could induce a high spatial variability of C stocks (Craft 2007; McLeod et al. 2011). Along the Baltic Sea, studies of carbon storage and sequestration in coastal wetlands are very rare.

Along the southern Baltic Sea, many coastal wetlands are dominated by *Phragmites australis*, which is principally adapted to freshwater conditions, but is able to cope with a wide range of salinities (Lissner and Schierup 1997; Engloner 2009; González-Alcaraz et al. 2012; Song et al. 2015). It has a very high biomass production that can lead to substantial soil C storage (Brix et al. 2001; Engloner 2009; Song et al. 2015). Moreover, the rooting depth of *Phragmites* is higher compared to other wetland species (Mozdzer et al. 2016). The enhanced sediment trapping by reeds would also support larger rates of vertical accretion compared to other wetland plant species (Clevering and Lissner 1999; Rooth et al. 2003). Yet, until now, there are relatively few studies addressing soil C stocks in reed-dominated coastal wetlands around the Baltic Sea (e.g., Callaway et al. 1996), as for other regions, too.

Measurements of organic carbon stocks in the sediment (0–1 m depth) at six representative locations in reed belts at the German Baltic Sea coast yielded values ranging from 8.3 to 37.7 kg C m^{-2} (Fig. 7.2). This concurs broadly with the global average for salt marshes, 25 kg C m^{-2} (Ouyang and Lee 2014). However, the variability within the sampled sites is high with coefficients of variations of about 50%.

Similar to the two main study sites, Dabitz and Michaelsdorf (see above), the six sampling sites differed with respect to salinity, the width of the Phragmites wetlands,

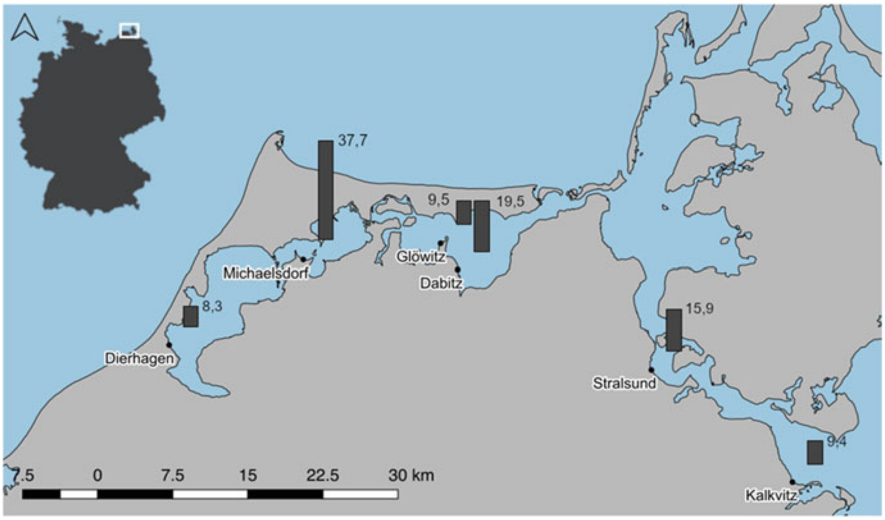

Fig. 7.2 Carbon stocks (kg m^{-2}) in sediments of the reed belts along the lagoons and estuaries along the Mecklenburg-Vorpommern Baltic Sea coast at six representative locations

topography, and land use in the hinterland. At Dierhagen, salinity is very low, and land use in the hinterland is pasture, but without a dyke. At Glöwitz, topography and land use are similar to Dabitz, but the reed belt is separated from the arable land by a strip of large trees.

The rates of carbon sequestration in the reed belt of the DZBC were estimated (at the Dabitz site) based on sedimentation rates which were determined at three sediment cores using the ^{137}Cs method, together with measurements of ^{210}Pb and Hg concentrations. The resulting sedimentation rates (for approximately the last 30 years) of 2–6 mm y^{-1} yielded carbon accumulation rates of 10–70 g C m^{-2} y^{-1}. This is relatively low compared with other coastal wetlands. The global mean for salt marshes is 242 g C m^{-2} y^{-1} (Ouyang and Lee 2014), whereas in the estuaries of the Oder and the Vistula, carbon sequestration rates of 100–400 g C m^{-2} y^{-1} have been determined (Callaway et al. 1996).

7.2.3 Coastal Wetlands as Buffers for Nutrients

Coastal wetlands can act as buffer and filter for nutrients, especially for phosphorus. Consequently, they are able to regulate the nutrient contents in the water body and counteract eutrophication (Karstens et al. 2015). The buffer and filter function of wetlands has proved so effective, that constructed wetlands are widely used in wastewater treatment facilities (e.g., Vymazal 2007; Koenig and Trémolières 2018). However, in natural coastal wetlands, the extent and processes of P retention and storage are different from those in constructed wetlands.

Phragmites australis is especially suitable to provide the ecosystem function of nutrient buffering: It has a very high biomass production that favors high soil C storage, trapping of sediment particles, and consequently accumulation and filtering of P (Brix et al. 2001; Engloner 2009; Song et al. 2015). It is able to cope with a wide range of salinities (Lissner and Schierup 1997; Engloner 2009; González-Alcaraz et al. 2012; Song et al. 2015), although it is principally adapted to freshwater conditions. Therefore, it is prevalent in different lagoon systems worldwide with very different salinities. In general, the function of coastal wetlands as a buffer and filter for nutrients, especially P, entails four main processes (cf. Reddy et al. 1999; Vymazal 2007) (Table 7.1):

- Sedimentation and peat accretion (sedimentation/resuspension).
- P sorption in sediment (adsorption/desorption).
- Chemical precipitation of Phosphate minerals (precipitation/dissolution).
- P uptake by plants (plant uptake/decay of dead plant material).

7.2.4 Sedimentation and Peat Accretion

As outlined above, coastal wetlands are often sites of active sediment accumulation, and in the DZBC, evaluation of historical aerial images suggests that *Phragmites*

Table 7.1 Processes and scenarios with potential for P removal from coastal wetlands in the DZBC

Process	P removal (g P m^{-2} y^{-1})	Assumptions	Evaluation
Sedimentation	0.52	Sedimentation rate 2 mm y^{-1}; BD 0.4 g cm^{-3}; P content 650 mg P kg^{-1} sediment	A part of the accreted P may be lost during diagenesis by dissolution and diffusion
	1.56	Sedimentation rate 6 mm y^{-1}; BD 0.4 g cm^{-3}; P content 650 mg P kg^{-1} sediment	
Adsorption	128	Under assumption of maximum sorption capacity	Short-term buffer
	0.1–12	Under measured ambient P concentrations in free water	
Plant uptake	Up to 8 (*Phragmites australis*)	Yearly harvest in late summer	This P amount could be removed entirely from the system

wetlands are able to keep pace with recent rates of sea level rise of 2.5–3 mm y^{-1} and may even prograding seawards. Based on a sedimentation rate of 2 mm y^{-1}, a bulk density of 0.4 g cm^{-3}, and a total P content of 650 mg P kg^{-1} sediment (Karstens et al. 2015), the yearly P accretion rate would be 0.52 g P m^{-2}, and 1.56 g P m^{-2} for the upper bound of sedimentation rates (6 mm y^{-1}). However, only part of the phosphorus trapped by this mechanism probably is removed from the system in the long term, whereas another part will dissolve in the pore water or be liberated by mineralization processes to diffuse upwards or will be consumed by plant uptake. Because the porewater P concentration generally exceeds the P concentration of the overlying water column, P is mostly transported from the sediment into the free water column by diffusive flux (Reddy et al. 1999). Moreover, plant demand for P (up to 8 g P m^{-2} y^{-1} for *Phragmites* at Dabitz) will be satisfied predominantly by P dissolved in the sediment pore space. Plant uptake of dissolved P likely entails a constant re-supply (by desorption and dissolution of phosphate minerals) from the sediment particles into the pore water solution.

Fluxes of dissolved P from sediments to the overlying water column are in general highly variable, with common values in the range of 0.02–3.2 mg P m^{-2} d^{-1} (Reddy et al. 1999) (corresponding to 7.3–1170 mg P y^{-1}). This suggests that an appreciable part of the P accumulated by sedimentation may be lost after sedimentation due to upward diffusion and plant uptake, however the exact proportion is not known. The reverse direction of transport—from the free water column into the sediment—is unlikely over the long term due to the low dissolved P concentrations in the free Bodden water (Berthold et al. 2018).

7.2.5 Phosphorus Sorption in Sediments

The sediments of the Phragmites reed belt in the DZBC have an exceedingly high sorption capacity for phosphate (Karstens et al. 2015). At the Dabitz site of the DZBC, phosphate sorption maxima (Langmuir equation) of about 16 g P kg^{-1} sediment were measured (Karstens et al. 2015) (0–10 cm depth, basin zone), whereas actual total P contents were merely 2.35 (0–2 cm depth) and 0.85 g P kg^{-1} (2–10 cm depth) sediment. These are extraordinarily high values for the sorption capacity, even when compared with previously reported values for sediments from the DZBC (Schlungbaum 1982b) and other areas of the Baltic Sea (Carman and Wulff 1989). They are probably caused by the high iron contents in this sediment of about 37 g Fe kg^{-1} sediment in 0–2 cm and 14.5 g Fe kg^{-1} sediment in 2–10 cm depth. Under natural conditions in the DZBC, only a small part of the sorption sites are thought to be occupied, whereas the maximum sorption capacity of 16 g P kg^{-1} corresponds to 128 g P m^{-2} (assuming that the upper 2 cm of sediment is in equilibrium with DZBC waters and a bulk density of the sediment of 0.4 g cm^{-3}). However, this large amount is merely a maximum theoretical value, which cannot be utilized under any naturally occurring conditions. Firstly, this maximum sorption capacity corresponds to equilibrium P concentrations dissolved in water of >100 mg P l^{-1}. Such concentrations are much higher than those observed in Bodden waters (well below 1 mg P l^{-1}). Using the Langmuir sorption isotherms determined for these sediments, the SRP concentrations commonly observed in Bodden waters (between 0.01 and 0.9 mg P l^{-1}) correspond merely to 0.14–11.7 g P m^{-2}. However, even these much lower amounts of P are probably in constant equilibrium with the Bodden waters, and therefore cannot be removed from the system in the long term. It can act as a short-term buffer, and a sink under certain conditions can very quickly turn into a source of P, if the physicochemical conditions (redox potential, turbulence, temperature, etc.) change.

Adsorption and desorption of phosphate are governed by the aeration status of the surface water, and consequently hydrodynamic and weather conditions: during windy weather with turbulent water flow and oxygenated surface water, iron is transformed into the oxygenated Fe(III) form (e.g., Fe_2O_3) with a high sorption capacity, whereas quiet weather conditions may lead to anoxic conditions in the surface water close to the sediment surface with iron in the reduced form Fe (II) which provides only few sorption sites for phosphate. These changes in the redox state can occur within hours (Karstens et al. 2015). At the reed belt site in Dabitz, concentrations of dissolved phosphate in the overlying water showed positive correlation with redox potential and dissolved oxygen content in the water and negative correlation with the water level (Karstens et al. 2015).

It is hardly possible to predict the direction of P flows, since the pertinent physicochemical conditions dependent on weather conditions cannot be predicted (Karstens et al. 2015). Similar restrictions presumably apply also to phosphorus minerals precipitated from the water. These are likely in equilibrium with the Bodden waters too, and therefore constitute a short-term buffer only. Therefore, in contrast to constructed wetlands in wastewater treatment facilities with high P concentrations in the wastewater, where P sorption has been identified as the quantitatively most important process of P removal (Vymazal 2007), this process has little significance for long-term P removal by coastal wetlands in natural lagoon systems with low P concentrations.

7.2.6 Phosphorus Uptake by Plants

Measurements during the course of a year revealed that at the Dabitz site, maximum phosphorus storage in the aboveground biomass of *Phragmites* amounted to up to 8 g P m^{-2} in the terrestrial and basin zone (Fig. 7.1), whereas maximum P storage in the fringe zone was somewhat lower (Berthold et al. 2018). This is largely within the range of reported phosphorus uptake capacities of emergent macrophytes in wetlands (Reddy et al. 1999; Vymazal 2007). However, P stocks in the biomass show a high temporal variability, with maximum values during the vegetation peak in August (Berthold et al. 2018). These large amounts of P are extracted during the vegetation period predominantly from the pore space of the sediment (Richardson and Marshall 1986; Reddy et al. 1999). After the peak in late summer, senescence of the *Phragmites* stands leads to a decrease in biomass and even more a marked decrease in aboveground P stocks, since P is to a large degree translocated from the aboveground parts into the rhizome system at the end of the growing season (Rodewald-Rudescu 1974; Schieferstein 1997).

Under undisturbed natural conditions, P stored in reed plants will be recycled annually within the system and therefore is stored only over medium terms (a few months). Unlike emergent vegetation, trees in forested wetlands possibly provide long-term P storage, with reported P uptake rates of 0.1–1.5 g m^{-2} y^{-1} (Reddy and DeBusk 1987). However, in coastal wetlands along the Baltic Sea, trees are rare. P stored in the aboveground parts of *Phragmites* stands can be largely removed from the system by reed harvesting. However, *Phragmites* harvest in northern Germany usually takes place in winter, when it does not interfere with nature-protection objectives such as bird protection and arrival of cranes in early autumn. Moreover, stems harvested in winter can be used directly as construction material without prior drying. However, P stocks of the aboveground *Phragmites* parts in winter are very low, and in order to remove the maximum amount of phosphorus from the system, harvest should ideally take place in late summer, in contrast to the current practice, if phosphorus removal is the goal. Maybe a feasible compromise between bird protection and nutrient removal could be a harvest in late September, when bird breeding might be over and phosphorus stocks in aboveground biomass would still be high (about 5 g P m^{-2}, Berthold et al. 2018).

To summarize, uptake by plants and export are the most important processes of P buffering in the long term, provided that the wetland belt is dominated by *Phragmites*, and the plants are harvested regularly in late summer. In contrast, P adsorption and precipitation are no effective mechanism for long-term P removal. However, they are important processes of P buffering on shorter timescales. Phosphorus accumulation by sedimentation and organic matter accretion in coastal wetlands are quantitatively important processes. However, the quantification of P accumulation rates is uncertain, because the determination of sedimentation rates is difficult (Nolte et al. 2013), and the P lost after sedimentation due to dissolution and diffusion out of the sediment and due to plant uptake is hard to measure (Reddy et al. 1999).

Sediment accretion rates, and therefore P accumulation rates depend on the type of hinterland, because topography and land use determine the amount of sediment that is transported into the coastal water body. In the DZBC, this can be clearly seen in the two study sites Dabitz and Michaelsdorf, as described at the beginning of this chapter (Karstens et al. 2016a). Since everywhere along the DZBC is mostly dominating, it can be assumed that the biomass of reed is comparable and consequently also the P extraction by reed plants.

Similarly to P accretion and buffering, the properties of the hinterland have an impact on carbon sequestration. GIS analysis of aerial images showed that the reed belts along the coasts of the DZBC have a total length of 194 km, an average width of 70 m, and cover altogether an area of 13.5 km^2. Assuming that 5 g P m^{-2} y^{-1} can be extracted by harvesting the reed annually in September (as discussed above), 67.5 t P per year could possibly be extracted from the system. That would be an appreciable amount which would probably have a quantitatively important impact on the DZBC. For comparison, the total P flux into DZBC is about 57 t per year (Selig et al. 2007), and the total content of soluble reactive P in the DZBC is about 5760 kg P (assumptions: volume 384×10^6 m^3 water with a mean concentration of 0.5 μmol l^{-1} PO_4, Schlungbaum et al. 1994). Recalculated for the catchment area draining into the DZBC of 1600 km^2 (Schlungbaum et al. 1994), this amount of P of 57 t would correspond to 0.42 kg P ha^{-1}, i.e., about the same amount of P which is lost by diffuse processes in the region (Umweltbundesamt 2017).

7.3 Seasonal Aspects and Short-Term Variability in Shallow Coast Ecosystems

In the shallow waters within the reed belts of the DZBC, several physicochemical parameters which have an impact on the phosphorus dynamics show a distinct temporal variation, both at seasonal and at shorter time scales (i.e., hours or days): water level, redox potential, oxygen saturation, temperature, pH. This is shown exemplarily in Fig. 7.3 for the dissolved oxygen concentration in the Basin zone at two different depths at the reed belt site Dabitz for a period of 7 months. There are distinct differences from day to day. Moreover, there is a seasonal variation with highest oxygen concentrations in the winter months.

Since oxygen and dissolved phosphate concentrations are (inversely) correlated (Karstens et al. 2015), these variations of oxygen concentrations at a fine temporal scale are an indicator of the temporal variability of dissolved P concentrations and the P dynamics in the shallow water of the reed belt. Significant changes of P dynamics can occur within hours accompanying a rapid change in weather conditions (especially wind speed and direction) and possibly strong local variations. This may be why correlations between the oxygen concentrations and dissolved

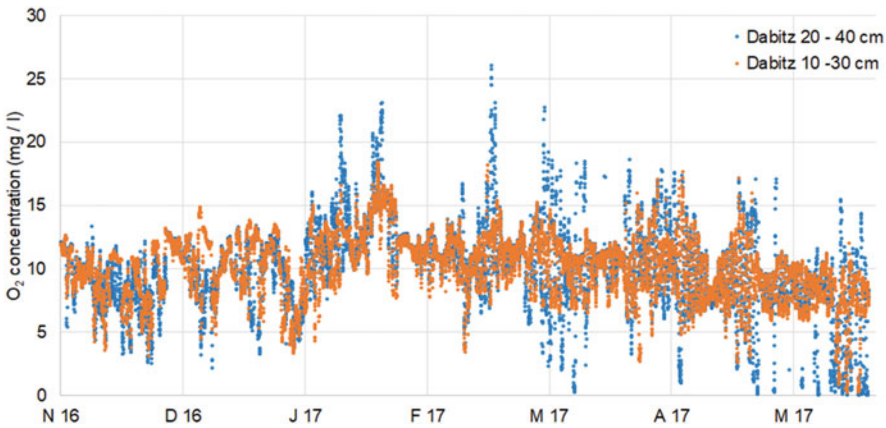

Fig. 7.3 Course of dissolved oxygen concentration at the Dabitz reed belt site, at the sampling location in the "basin zone" of the reed belt, two different water depths; about 20–40 cm (dependent on temporally varying water level) and about 10–30 cm at the same location, directly above

Table 7.2 Oxygen saturation and SRP concentrations in the water of three locations in the basin zone at Dabitz for two events with rapidly changing weather and hydrodynamic conditions (24/25th of June 2014 and 8th July 2014) (Karstens et al. 2015)

	Landward side		Basin center		Seaward side	
	O_2-saturation (%)	SRP (mg l^{-1})	O_2-saturation (%)	SRP (mg l^{-1})	O_2-saturation (%)	SRP (mg l^{-1})
2014-06-24 21:30	0.9	0.129	15.9	0.037	9.2	0.084
2014-06-25 07:30	4.1	0.107	37	0.023	72.2	0.043
2014-07-08 11:00	3.8	0.029	62.1	0.011	11.6	0.012
2014-07-08 18:00	90.5	0.013	111	0.007	49.5	0.006

phosphate are rather weak ($R^2 < 0.6$), and the phosphate concentrations depend on more factors, so that the oxygen saturation cannot be used directly as a sole proxy for dissolved P concentrations.

The rapidity of changes in both oxygen concentrations and dissolved P concentrations is illustrated in Table 7.2 for two events of rapidly changing weather conditions at the Dabitz site (Karstens et al. 2015): for both events, oxygen concentrations in the water rise distinctly within hours, due to a rapid change of wind directions (from low wind conditions towards strong northeastern winds). Concomitantly with this rise in oxygen saturation, there is a rapid decrease in dissolved phosphate concentration.

7.4 Long-Term Trends of Shallow Coast Ecosystems

As discussed above, coastal reed belts and brackish grasslands can counteract rising sea levels by vertical accretion. This adaptive capacity depends on anthropogenic influences, among others, the type of land use in the hinterland and coastal protection dykes, because these factors determine the amount of sediment that is delivered into the wetland. This in turn has important ramifications for the ability of the coastal wetland for accretion and therefore to accommodate rising sea levels and to sequester organic carbon.

The two intensively studied sites at the DZBC, Dabitz and Michaelsdorf, are representative for two contrasting land use cases in the hinterland and can therefore be used to study the impact of different land use on reed belt development: the Phragmites wetland at Dabitz borders directly on cropland, whereas the wetland at Michaelsdorf is “squeezed” behind a dyke and the hinterland used as pasture for sheep (Karstens et al. 2016a).

At Michaelsdorf, analysis of aerial images shows that the wetland extent decreased by about 25% after the dyke construction which took place in the early 1970s (Fig. 7.4, area in 1953: 67,404 m^2; area in 1983: 50,425 m^2). Between 1983 and 2000, the wetland expanded again seawards by 16% and the analyzed wetland

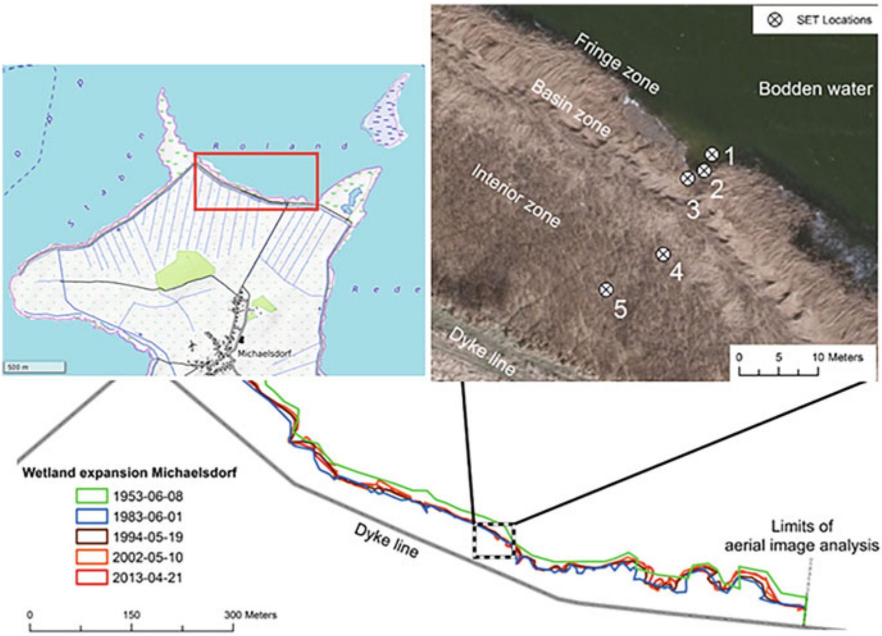

Fig. 7.4 Location of the analyzed reed belt north of Michaelsdorf in the Darss-Zingst Bodden Chain (red rectangle) (top left); location of five permanent measurement locations for the Surface-Elevation Table (top right; Karstens et al. 2016a); development of the seaward boundary of the *Phragmites* wetland in Michaelsdorf between 1953 and 2013 derived from aerial images (bottom)

area reached 61,461 m^2, followed by a slower retreat since 2000 (56,191 m^2 in 2013). In 1953 the wetland edge was located up to 10 m further into the water than in 2013. In 2000, the peak of the recovery phase after dyke construction, the wetland edge was located 5 m further into the sea than in 2013.

Surface elevation table (SET) analyses may help to understand the long-term potential for surface accretion from short-term measurements. We used SET measurements, i.e., measurements of the relative height of the surface and its temporal development to unravel the recent dynamics in the reed belt of Michaelsdorf from March 2014 until March 2015 (Karstens et al. 2016a). These measurements were performed at five spots in the reed belt (Fig. 7.5). Whereas at none of those measurement locations a vertical accretion during the measurement period was observed, two spots ("micro cliff" and "interior zone a") revealed an overall lowering of the surface (either by erosion or subsidence) of more than 3 cm during this single year (Fig. 7.5). Since the spot "micro cliff" is located directly at the seaward boundary of the reed belt, this lowering translates directly into a receding shoreline. Our results suggest that none of the SET measuring locations can currently keep up with the local sea level rise.

To summarize, both analyses of historical aerial images (since 1953) and fine-scale measurements of short-term vertical land-surface movements (in 2014) (by means of SET measurements) indicate that those parts of the reed belt are threatened which are confined by a dyke to the hinterland and do not receive sediment particles from it, due to the flat topography, the land use, and the dyking. In the DZBC, the coast is dominated by such conditions and thus the development of the reed belt faces the threat of "coastal squeeze" (Doody 2013). This means, vertical

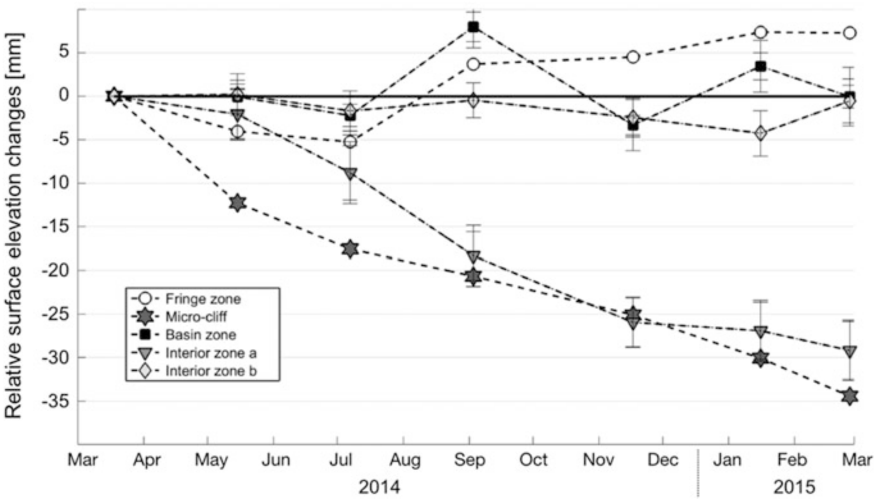

Fig. 7.5 Surface elevation changes [mm] over time for the five positions at the study site (numbers according to Fig. 10.10: 1, fringe zone; 2, micro cliff; 3, Basin zone; 4, interior zone a; 5, interior zone b). Symbols represent the means of the 60 pins and error bars the standard error

accretion of the reed belt cannot cope with the rising sea level, so that the reed belt retreats seawards, whereas it is restricted landwards by the dyke. This applies to a large degree to the study site Michaelsdorf. A remedy against such a scenario could be the abandonment of the present dyke line and the area currently used as grassland.

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Benthic Habitats and Their Inhabitants

8

Michael L. Zettler and Alexander Darr

Abstract

In this chapter, we describe the formation and mapping of marine benthic underwater habitats in offshore waters. We characterize the respective biotopes and discuss the anthropogenic pressures acting upon them. A variety of different classification systems have been developed within the last decades that, depending on the respective aim and scale, significantly differ in complexity and splitting rules. An accepted system has been developed and adapted to specific conditions in the Baltic Sea. The advantage of the so-called HELCOM Underwater Biotope and Habitat Classification System (HUB) is the clear definition of splitting rules between the different modalities within all six hierarchical levels. Despite the fact that some of them could be the result of artificial separations due to the different hierarchical systems, we have found an impressive expression of the biotope diversity in the southern Baltic Sea and its ecological potential.

8.1 Genesis

The southern Baltic Sea can roughly be divided into coastal inshore and coastal offshore waters. The habitats of the inshore ecosystems with lagoons, estuaries and bays and its hinterland are elaborated in the previous parts of Chap. 3. Thus, we focus in the following on the offshore coastal system (territorial and EEZ) of the German Baltic Sea and here on the benthic compartment. The benthic environments in the Baltic are often considered to consist of homogeneous sand and mud which are almost deserted from benthic life. But, also the Baltic accommodates a fascinating

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diversity of habitats. Whereas a habitat is defined as the abiotic environment that contributes to the nature of the seabed, a biotope is defined as the combination of a habitat and its associated community of organisms exhibiting a distinct community function (Avellan et al. 2013a).

The habitat diversity has its seeds in the genesis of the Baltic Sea. Analogous to the hinterland, the last glacial periods had a vast terraforming impact on the underwater landscapes. Moving ice shields and melting water run-off formed moraines, dunes, basins, drowned riverbeds and deltas. These primary landscapes were subsequently reworked and diversified by various environmental factors forced by climatic conditions and changes in seawater level (Chap. 3). Water currents driven by wave energy or internal circulation play a major role for erosion and sorting of the sediments. While it is true that wide flat areas are dominated by homogeneous sand, the moraines still exist. They are characterized by glacial till comprising a mixture of different substrates frequently including cobbles and boulders. Where the sand is elevated (e.g., sunken dunes or physically accumulated), sandbanks emerge providing a variety of microstructures for benthic biota.

While these elevated banks are similarly to the shallow areas along the shoreline almost permanently exposed to wave energy resulting in a permanent re-organization of sediment, the physical stress is reduced with increasing water depth. Consequently, the basins act as sedimentation “traps” especially for smaller grain sizes such as silt, but also for organic material originating from the water column. The topography of the Baltic with sills, moraines, and basins has a strong impact on the water circulation and exchange. This leads to strong gradients in oceanographic parameters such as salinity, temperature, and oxygen supply that again have a strong impact on the benthic inhabitants and lead to diversification of communities (Zettler et al. 2014).

8.2 Detection and Mapping

The adequate method for detection and mapping of habitats strongly depends on the water depths. While on land, biotope mapping often consists of a combination of on-site inspections and digitalized remote sensing methods from the air (drones or planes) or space (satellite), this approach is only applicable in very shallow parts of the sea. Depending on the visibility of the water mainly geological structures, but also macrophyte meadows or mussel banks can be identified and mapped. In deeper waters, no area-wide optical survey is possible. Here, geophysical surveys using different hydroacoustical methods are the basis to identify sediment composition and topographical features. The acoustical backscatter has to be interpreted and transformed into the relevant sediment class. This is done by means of physical sediment samples and optical information gained either by towed underwater videos or by remotely operating vehicles (ground truthing). However, this procedure is time-consuming and has not yet been finalized for large parts of the south-western Baltic Sea. Consequently, in large areas habitat information still bases on the spatial interpolation of sediment information physically sampled over a large time period

(Tauber 2012). Also the biological information to raise the level of detail from habitat to biotope mapping currently bases on the extrapolation of stations data mainly gained by grab sampling (e.g., Darr et al. 2014; Schiele et al. 2015).

8.3 Classification

Ecosystem-based management demands a clear identification and separation of the diversity of underwater marine habitats that often contradicts both the smooth transitions as well as the temporal variability in nature. A variety of different classification systems have been developed within the last decades that, depending on the respective aim and scale, significantly differ in complexity and splitting rules. While the European habitats directive only defines specific large-scale biotope complexes, the European nature information system (EUNIS) defines several levels heading from broad habitat types defined by geological and physical key parameters towards biotopes dominated by specific communities. A comparable system was developed and adapted to specific conditions in the Baltic Sea. The advantage of the so-called HELCOM Underwater Biotope and Habitat Classification System (HUB, Avellan et al. 2013b) is the clear definition of splitting rules between the different modalities within all six hierarchical levels. Schiele et al. (2015) generated a first area-wide biotope map of the German part of the Baltic Sea. A re-calculation of the map based on more actual data and including a slight adaptation of the modeling procedure in general confirmed the results (Fig. 8.1, Table 8.1). However, the

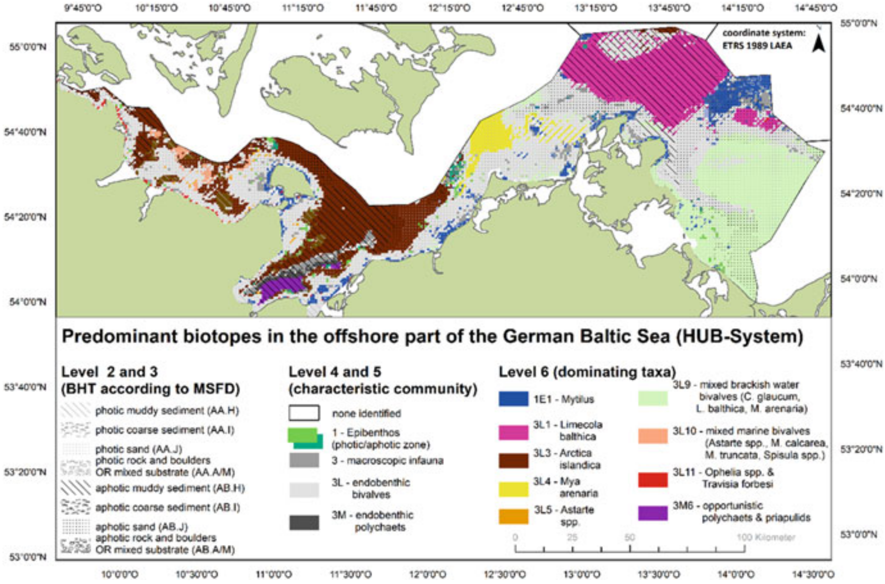

Fig. 8.1 Predominant biotopes in the offshore part of the German Baltic Sea using the HELCOM Underwater Biotope classification (data status: 2018)

Table 8.1 Coverage area of some main offshore habitats occurring in the German Baltic Sea (aphotic and photic areas are combined)

Habitat	Dominating taxon	Coverage area (km^2)
Mud		**4033**
	Arctica islandica	1300
	Limecola balthica	1600
	Others/unknown	1133
Sand		**9582**
	Arctica islandica	1030
	Mya arenaria	295
	Brackish water bivalves	2452
	Others/unknown	5805
Coarse substrate (gravel, pebbles)		**98**
Mixed substrates including glacial till, cobble, and boulder fields		**529**
Other (e.g., clay, peat)		**35**
Eelgrass meadows	*Zostera marina*	**180–203**[a]

[a]Estimate based on Schubert et al. (2015), Schubert and Schygulla (2018) and Bobsien et al. (2020)

inclusion of an evaluation of confidence reveals that in quite large areas, the actual dominating community is not known. Especially in shallow waters along the coast, substrate heterogeneity and the presence of communities not yet considered in the HUB system prohibit a clear identification of biotopes. Also along smooth gradients of the driving environmental parameter, communities overlap and a clear separation is not always possible. A prominent example is the decline in abundance of the lagoon cockle *Cerastoderma glaucum* and the sand gaper *Mya arenaria* with increasing amount of organic material in the sediment along the depth gradient from Pomeranian Bay towards Arkona Basin. The decline of these species leads to a gradual replacement of the mixed brackish water bivalve's community (HUB L9) by a community solely dominated by the Baltic tellin *Limecola balthica*. But also temporal variability adds uncertainty to the map. In the Arkona Basin, the Baltic tellin currently is the dominant macrobenthic species in terms of biomass. But frequently, larvae of the ocean quahog *Arctica islandica* are introduced into the area from Øresund. However, up to now, the species does not succeed in building up an autochthonous population in main parts of the basin. The specimens disappear after growing for a few years. This implicates a decrease in confidence in parts of the Arkona Basin (pink hatched areas).

Besides these predominant biotopes, the south-western part of the Baltic Sea features a variety of biotopes with limited size and/or distribution. Often they are characterized by individual key structure-forming species such as blue mussels *Mytilus* spp., common eelgrass *Zostera marina* or bladder wrack *Fucus vesiculosus*. The Baltic Sea Red List of biotopes identifies around 300 different benthic biotopes (Avellan et al. 2013a). Despite the fact that some of them might be the result of artificial separations due to the distinct hierarchical system, this figure is an impressive expression of the biotope diversity in the Baltic Sea and its ecological potential (Fig. 8.1).

8.4 Anthropogenic Pressures and Conservational Aspects

The marine sublittoral benthic environment does not only form important ecological habitats but also supply essential functions to the entire marine ecosystem. For example, they act as settling grounds, feeding, and nursery areas for diverse sessile and mobile marine species and communities. This functioning might suffer from different stressor arising from human activities. Both physical (e.g., bottom trawling) and chemical (e.g., eutrophication) disturbances cause widespread impacts on marine ecosystems, changing the general characteristics of the seabed and their associated benthic invertebrate communities (for references, see van Denderen et al. 2019). In a fragile ecosystem such as the Baltic Sea, which already suffers from natural stress, large-scale impacts potentially affect entire habitats and the ecosystem. For example, bottom trawling might be highly relevant for our study area (ICES 2017), where otter trawls target demersal fish such as cod, plaice, or flounder. Additionally, installations for offshore wind farms, cable, or gas pipelines affect benthic habitats temporarily or permanently on a local scale. The impacts of more diffuse stressors such as pollution, marine litter, and neobiota on benthic habitats and their inhabitants are poorly investigated. However, as the ecosystem function and its provided services (Chaps. 20–26) are not only ecologically but also economically relevant, European conservation and water policies have raised the importance of a healthy ecological state.

The Habitats Directive is one of the main legal tools of the European Union to preserve biodiversity by maintaining and restoring natural habitats, and establishing a network of protected sites (special areas of conservation SAC, European Commission 2013). While the habitats directive focuses on particular important biotope complexes, red lists take a closer look on the current state of threat of all relevant biotopes on national scale (Fürhaupter et al. 2017), on the Baltic scale (Avellan et al. 2013a) as well as on European scale (Gubbay et al. 2016). Finally, the actual state of all natural habitats and biotopes is assessed under the Marine strategy framework directive (MSFD) with the goal to maintain or restore vital marine habitats. In this process, high-resolution knowledge on the distribution and biological constitution of benthic biotopes are the basis for the implementation of purposeful measures such as the identification of functional and biodiversity hotspots, areas of specific sensitivity against specific pressures and consequently the identification of potential additional marine protected areas and the elaboration of management plans.

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Part III

Ecological Structures and Functions of Coastal Water and Offshore Ecosystems

Introducing the Ecological Aspects

9

Irmgard Blindow and Stefan Forster

Abstract

Combined results from both the BACOSA and SECOS projects highlight structure and function of aquatic Baltic Sea ecosystems in a gradient from land to open Sea.

Coastal lagoons differ highly not only in their hydrological and nutrient regimes: Lagoons with dense submerged vegetation display higher biomass and species richness at all trophic levels and reveal higher trophic transfer efficiency than more eutrophicated lagoons with sparse vegetation. In the Darß-Zingst Bodden chain, high-resolution and long-term data identify the importance of extreme, but rarely occurring events for changes in abiotic and biotic parameters, such as oxygen depletion under the ice cover or Major Baltic Inflows. The Vitter Bodden is characterized by dense submerged vegetation and high food web connectivity, but low recycling, redundancy and short trophic pathways indicate that this system might be close to the "tipping point" and at risk for drastic changes.

Based on aggregated data, benthic faunal communities and their traits in the offshore waters are mapped and hot spots of functional aspects are identified, supporting projections of marine ecosystem features. We compare bioturbation values measured directly with those captured by corresponding indices, in an attempt to highlight this functional aspect of the benthos. Seasonal dynamics of plankton communities differ markedly between Belt Sea and Baltic Proper. Our data show how understanding of short-term anomalies and long-term variability is important for assessing implications of climate change. Long-term data sets in

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the pelagic and benthic realm play therefore a crucial role in assessing the state of the marine systems and changes, such as regime shifts and longer growing seasons.

Long-term monitoring data from inner and outer coastal waters identify six possible phytoplankton blooming types and different spatio-temporal limitation regimes, revealing rather stable intra-annual patterns, but almost no long-term trends. This indicates a high system resilience, which may be favourable to sustain a certain ecosystem state, but unfavourable if such systems need to be restored.

Coastal ecosystems are of high importance for recreation and economy, but simultaneously exposed to a number of environmental stressors such as eutrophication, climate change, over-exploitation and invasive species. The projects BACOSA and SECOS aimed at evaluating historical changes of the inner and outer coastal waters, investigating the complex interactions between abiotic parameters and organisms and finally, linking these empirical results to ecosystem services (see Chap. 28). This chapter describes and analyses structural and functional aspects of inner and outer coastal water as well as offshore ecosystems of the southern Baltic Sea. After presenting the classification of different ecosystem types (Chap. 10), the following sections deal with inner coastal ecosystems, focusing on two coastal lagoon ecosystems, which were intensively studied during the BACOSA project. After a short comparison of abiotic parameters and primary producer dominance between both lagoon systems (see Chap. 11), results derived from highly diverse data sets are presented. Long-term data with high resolution are available for the Darß-Zingst Bodden chain and enable to identify and quantify the impact of various "extreme events" on physico-chemical parameters, biotic components and finally, ecosystem services provided by this ecosystem (see Chap. 12). Intensive investigations of all food web compartments in the Vitter Bodden and the Grabow illustrate the major differences of carbon fluxes and trophic interactions between two lagoon systems (see Chap. 13). Starting with Chap. 14 the focus changes to the offshore ecosystems. The ecological structure in benthic habitats is presented (Chap. 14) based on long-term data, followed by a discussion of some traits and a functional aspect, the community bioturbation potential. Based on targeted studies in SECOS, particle reworking and bioirrigation are more specifically addressed (Chap. 15), and a potential effect on down-slope particle transport is hypothesized. Chapter 16 presents seasonal aspects and short-term variability in the pelagic system showing that these indeed reflect the differences in environmental conditions between Belt Sea and southern Baltic Proper. Chapter 17 depicts trends and regime shifts in the southern Baltic Sea, both in the pelagic and benthic system, addressing a temperature effect and the elongation of the pelagic growing season.

A final synthesis combines inner and outer coastal waters. It identifies different patterns of seasonal phytoplankton development and periodicities in nutrient limitation regimes comparing coastal lagoons and a number of close by outer coastal water bodies (Chap. 18).

Baltic Sea Aquatic Ecosystems in a Gradient from Land to Open Sea

10

Irmgard Blindow, Maximilian Berthold, Stefan Forster, and Hendrik Schubert

Abstract

Situated in a gradient from land to the open sea, coastal water bodies can be separated into outer coastal waters and inner coastal waters. The southern Baltic Sea shows mostly lagoon-like coastal waters. Though connected to the open sea, such coastal lagoons are protected by land splits or similar land features and therefore highly exposed to influences from terrestrial ecosystems, especially nutrient runoff causing eutrophication. In the confusing complexity of different, often regional names for these ecosystems, we here use a terminology originating from the international literature. 'Estuarine lagoons' are inner coastal water bodies with high contribution of terrestrial inflows and "marine lagoons" are inner coastal water bodies without a major freshwater input, but high water exchange with the open Baltic Sea.

Coastal water bodies are the first aquatic ecosystems in a gradient from land to open sea. They can be separated into ecosystems along the outer coast and *inner coastal waters* (= *coastal lagoons*). Coastal lagoons are according to a formal definition (United Nations 1997) "seawater bodies situated at the coast, but separated from the

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sea by land spits or similar land features. Coastal lagoons are open to the sea in restricted spaces." Morphology, hydrology, tidal, and salinity characteristics differ largely among coastal lagoons, also called inner coastal water bodies or estuaries (Tagliapietra et al. 2009). The terminology used for these habitats is confusing: different names are used in different regions and by different authors (Tagliapietra et al. 2009).

The Baltic Sea includes regions with a straight coastline, and regions with coastal lagoon systems, all of which microtidal and brackish due to the nature of the Baltic as an inland sea. Far different terminologies have been used to classify Baltic Sea lagoons. In the northern part of the Baltic Sea, land uplift is still high and causes a gradual change of coastal waters from open lagoons to water bodies with an increasingly restricted water exchange with the open Baltic. A specific terminology has been developed for this region, which distinguishes the different stages of this temporal development (Munsterhjelm 1997).

Along the southern coast of the Baltic Sea, post-glacial rebound is negligible. The morphology of the lagoons is therefore not exposed to equally drastic changes as described for the northern Baltic Sea (Munsterhjelm 1997). The region is characterized by a pattern of eroding Holocene cliffs and depositional areas, which creates a chain of semi-enclosed inland water bodies. Some of these lagoons, especially the Schlei, the Darß-Zingster-Boddenkette and the Nordrügenschen Bodden, Schlei, are complex systems consisting of interconnected shallow water bodies, whereas the Wismarbucht, Salzhaff, and Greifswalder Bodden are characterized by a rather simple morphology. These transitional zones between the mainland and the open Baltic Sea host a rich fauna and flora, serve as spawning and nursery grounds, and provide feeding areas for a multitude of species (see Chap. 13). They are a region with high anthropogenic pressure and provide numerous ecosystem services (see Chap. 28).

Regionally used classifications of southern Baltic Sea lagoons have mainly considered both the water exchange with the open Baltic Sea and terrestrial inflows, with respect to the fact that the relative contribution of both is crucial to the nutrient status of the lagoons. These classifications are based on the influence of hydrology on the functioning of these coastal lagoons (Schlungbaum and Baudler 2000, 2001; Schiewer 2008). The parameters chosen for typology reflect the combined effects of main determinants for hydrological conditions as, e.g., ratio of water body surface to catchment area surface, ratio of coastline to water body surface, water exchange rates, and precipitation. Combined with nutrient budgets from riverine input and groundwater, this information gives an estimate of nutrient discharge into the lagoons and allows descriptions of maximum allowable nutrient inputs. For the German Baltic coast, inner and outer coastal water bodies were sorted according to their salinity into groups ranging from oligo- (salinity <5) to mesohaline (salinity 5 to <18) (Sagert et al. 2008). The FFH habitat directive distinguishes among habitat types 1130, Estuaries, 1150, Coastal lagoons and 1160, Large shallow inlets and bays (EC 2003).

Different hydrodynamic exchange regimes cause various shares of water exchange with the open Baltic Sea and terrestrial runoff among coastal lagoons, causing not only differences in salinity patterns, but also differences in water column nutrient concentrations. To a varying extent, all coastal lagoons are exposed to eutrophication caused by terrestrial inputs, and act as nutrient filters between land and open sea, thereby protecting the sea from eutrophication (Asmala et al. 2019; Carstensen et al. 2020). During the last decades, eutrophication resulted in a deterioration of lagoon ecosystems and created a demand for sustainable management plans. The HELCOM Baltic Sea Action Plan (HELCOM 2007; Backer et al. 2010), an ambitious program, aims at restoring the good ecological status also of coastal lagoons.

The main result of our detailed analysis of two lagoon ecosystems is that a common type of southern Baltic Sea coastal lagoon does not exist, but that these lagoons can be totally different in terms of species composition, biomasses, and production of single functional groups, as well as food-web structure and functioning. These differences can ultimately be attributed to the hydrology patterns of the lagoons, why we feel a necessity to use different terms for these lagoon types. Instead of applying any local or regional terminology (see above), we follow the detailed analysis of Tagliapietra et al. (2009) and here, use the terms "*estuarine lagoon*" for an inner coastal water body with high contribution of terrestrial inflows and "*marine lagoon*" for an inner coastal water body without a major freshwater input, but high water exchange with the open Baltic Sea. We neglect, however, Tagliapietra's additional criterion of salinity differences between estuarine and coastal lagoons, as the Baltic Sea in its whole is a brackish water ecosystem.

The term *outer coastal waters* is used for the reach within one nautical mile off the outer coastline. This complies with the Water Framework Directive, which defines coastal waters as surface waters within 1 nm off the outer coastline when attributing the ecological status, but within distances of up to 12 nm when assessing the chemical status. Given fewer geomorphological restriction, outer coastal waters are clearly distinguished from coastal lagoons by more horizontal mixing, frequently less intense eutrophication and often higher salinities. However, and differing from most marine realms, the Baltic itself displays pronounced salinity gradients and fluctuations (Bleich et al. 2011) and a generally elevated eutrophic level.

The use of the terms *offshore* (versus *inshore*), *onshore*, *sublittoral* or *coastal* varies in and among marine science fields, marine conservation, fishing, oil exploitation, and other marine uses. The term *offshore* is frequently applied in the context of shelf seas and the open ocean dozens of nautical miles from any coast and water depths exceeding dozens of meters. In this contribution, we use the term *offshore* for ecosystems seaward of the coastline, including the outer coastal waters, despite the fact that in the Southwestern Baltic those locations have rather limited water depths and distances from the coast. In the Baltic Sea this implies a gradient from permanent stratification distant from to weak stratification close to the coast. Offshore coincides with the Exclusive Economic Zone (EEZ) in this area (Fig. 10.1).

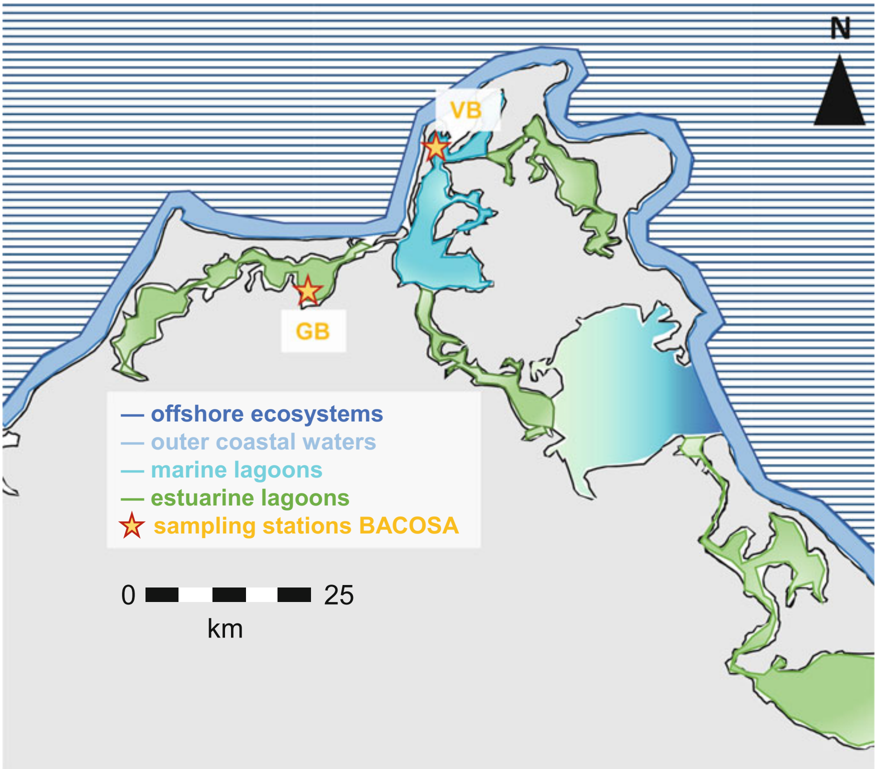

Fig. 10.1 Different types of Baltic Sea aquatic ecosystems. Asterixes show the coastal lagoon sampling stations Grabow (GB; Darß-Zingst Bodden chain) and Vitter Bodden (VB; Westrügensche Bodden)

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11 Comparison of Abiotic Parameters and Dominant Primary Producers Between the Two Main Investigation Areas

Rhena Schumann and Irmgard Blindow

Abstract

The Southern Baltic Sea coast is dominated by lagoons, which are called Bodden and Haffe in relation to the geological background and their formation. They are all heavily eutrophicated, but differ in the eutrophication grade by their catchment area, water retention and exchange with the Baltic. Here, two contrasting Bodden lagoons will be compared. One is the more eutrophicated Darß-Zingst Bodden with a large catmint from agricultural use and a small exchange to the Baltic versus the Vitter Bodden with a smaller catchment and a much larger exchange to the Baltic by two openings. The differences in morphology and hydrology have a tremendous effect on all water parameters, which result in completely different macrophyte colonization.

Intensive food web investigations within the BACOSA project were focussed on two coastal lagoons, the estuarine Darß-Zingst Bodden chain, specifically the site Grabow, and the marine lagoon Westrügensche Bodden, specifically the Vitter Bodden (see Fig. 10.1).

Both lagoons have a similar size and mean depth and are situated within the same geographic region, but differ widely in nutrient conditions and biotic components (see Chaps. 12 and 13). The Darß-Zingst Bodden has a far larger catchment area (Table 11.1) and is more eutrophicated than the Vitter Bodden, which is indicated by higher seston, chlorophyll and nutrient concentrations and lower light availability in the water column (Table 11.2).

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Table 11.1 Hydrological characteristics of both investigated coastal lagoons

	DZBC	WRB
Area [km^2]	197	171
Volume [10^6 m^3]	397	300
Mean depth [m]	2.0	1.8
Catchment area [km^2]	1594	238
Water surface: catchment area	1:8	1:1
Freshwater inflow [10^6 m^3 a^{-1}]	290	n.a.
Baltic Sea inflow [10^6 m^3 a^{-1}]	2750	n.a.
Outflow [10^6 m^3 a^{-1}]	3020	n.a.
Water exchange rate (times a^{-1})	0.13	n.a.

Sources: Schlungbaum and Baudler (2001), Schiewer (2008)
DZBK Darß-Zingst Bodden chain, *WRB* Westrügensche Bodden, *n. a.* data not available

Table 11.2 Environmental parameters of Grabow (above) and Vitter Bodden (below) measured in 2017

Month	Salinity	Seston mg L^{-1}	Chl *a* µg L^{-1}	Secchi m	KD m^{-1}	TN µM L^{-1}	TP µM L^{-1}
March	9.1	17.3	14.1	1.5	0.7	54.9	1.6
April	9.6	8.2	7.8	1.8	0.5	45.9	0.9
May	8.5	34	42.9	0.5	1.6	68.4	1.9
June	9.2	96	55.9	0.3	2.3	148.9	4.4
July	8.3	40	26.5	0.5	1.4	94.0	3.4
August	8.5	33.1	31.5	0.5	1.5	109.2	2.8
September	8.5	15.7	15.6	0.8	1.1	124.5	2.3
October	7.8	69.5	69.5	0.6	1.7	135.0	3.8
November	8.4	21.9	21.9	0.6	1.4	68.8	2.0
December	8.2	14.8	14.8	0.7	1.4	228.2	2.5
Month	Salinity	Seston mg L^{-1}	Chl *a* µg L^{-1}	Secchi m	KD m^{-1}	TN µM L^{-1}	TP µM L^{-1}
March	9.5	7.7	1.5	2.6	0.2	35.5	0.7
April	9.3	7.7	0.7	2.4	0.3	40.7	0.9
May	9.7	6.4	2.4	2.8	0.2	31.1	1.1
June	9.5	13.0	10.5	1.6	0.7	48.1	1.1
July	9.6	12.4	6.7	2.2	0.5	30.3	1.4
August	9.1	5.8	5.8	2.2	0.6	42.8	1.2
September	8.8	3.8	4.2	2.4	0.4	n.a.	n.a.
October	9.0	3.5	1.9	2.4	0.4	n.a.	n.a.
November	9.9	10.6	2.5	1.4	0.6	44.1	1.2
December	9.6	8.7	3.3	2.0	0.5	n.a.	n.a.

From Paar et al. (2021), modified
Chl a the chlorophyll *a* concent, *Secchi* the Secchi depth and Kd the light attenuation coefficient, calculated out of the slope of log PAR with increasing water depth, *n.a.* data not available

11.1 Darß-Zingst Bodden Chain (DZBC)

The DZBC consists of four consecutive water bodies with a highly restricted water exchange not only between the lagoon chain and the Baltic Sea, but also between the single adjacent basins (Schumann et al. 2006). Consequently, there is a strong salinity gradient with increasing salinities from the innermost to the outermost basin, and an equally strong nutrient gradient with the highest loads and concentrations in the innermost parts (Schumann et al. 2001). These gradients have a major impact on phytoplankton composition and production (Wasmund 1990; Schoor et al. 2008).

A drastic decrease in macrophyte biomasses by 30–70% including a shift in species composition was observed between the 1970s and 1994 following elevated external nutrient loadings. While charophytes were abundant in former times, the submerged vegetation is today dominated by vascular plants such as *Stuckenia pectinata*, *Ruppia* spp. and *Zannichellia* sp. (Lindner 1972; Festerling 1973; Behrens 1982; Teubner 1989; Schiewer and Schumann 1994).

The sampling site Dabitz is located at the western shore of the Grabow (GB), the outermost basin.

11.2 Westrügensche Bodden (WRB)

The WRB consists of the basins Kubitzer Bodden, Schaproder Bodden and Vitter Bodden (from south to north), the latter one containing the BACOSA sampling site VB. The WRB has two connections with the open Baltic Sea, one in the SW and the other one in the NE. Data on water exchange with the open Baltic Sea are not available, but periods of drastic water level changes cause distinct inflow or outflow events. As all of the few freshwater inflows are small, the water volume added by these inflows is probably low (data not available).

The WRB is characterized by dense submerged vegetation, reaching down to more than 4 m and covering the major part of the sediments (Blindow et al. 2016; Bühler 2016). While the cover and depth distribution of this vegetation hardly has changed at least since the 1930s, a major shift in species composition occurred from low-growing “bottom-dwellers” to taller species (Blindow et al. 2016).

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Short-Term Variability, Long-Term Trends and Seasonal Aspects in the Darß-Zingst Bodden Chain

12

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Abstract

This chapter concentrates on one of the most eutrophicated inner coastal waters of the German Southern Baltic, the Darß-Zingst Bodden in contrat to other more open coastal areas (e.g. Feuerpfeil et al. Estuar Coast Shelf Sci 61:89 -100;2004). Similarly eutrophicated lagoons are the Northern Rügensche Bodden, i. e. the Jasmunder Bodden. Most of the other lagoons along the Southern Baltic coast have a smaller catchment and a higher exchange with the Baltic. The Darß-Zingst Bodden was investigated for 50 years now by state authorities and the University of Rostock with more and more parameters adding up by the years. Online probes were installed in the early 2000s to evaluate diurnal, seasonal and annual variability.

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12.1 Introduction

The Darß-Zingst Bodden chain (DZBC) is a densely monitored estuarine lagoon system with first scientific descriptions starting in the 1930s. Since the Biological Station of the University of Rostock started in Zingst, regular state monitoring can be combined with high-frequency monitoring and research. Weekly measurements of abiotic parameters were gradually extended to biotic parameters and daily measurements. With the availability of online-logging systems, high-frequency data with 10-min intervals could be obtained. Research vessels allowed in-depth analyses of specific research questions such as nutrient cycles, as well as spatial monitoring along the complete lagoon gradient. Such high-resolution data can be used both to identify long-term ecosystem changes and to evaluate the impact of short-term events, like hypoxia, elevated precipitation or invasion of new species. Here, we present and discuss the effects of some noteworthy events that affected the ecosystem during the last decades.

12.2 Data Overview

1D-high resolution data (every 5–10 min) suitable to evaluate short-term variability and to uncover extreme values of, e. g. oxygen depletion, water mixing by salinity changes were recorded at one central site. The nutrient and salinity gradient within the DZBC is sampled monthly at nine stations, if possible also in winter. For other inner coastal waters, data from state agencies or publications are available, but mostly only from the vegetation periods. One large data set is of course the DWD (German weather service) with precipitation, radiation and temperature data. Additionally, there is a continuous monitoring of air quality by the UBA (Umweltbundesamt) located near Zingst. Within the BACOSA project, in-depth investigations were conducted for two very different sites in the years of 2014 and 2017 to investigate more compartments as well as food webs.

12.3 Long-Term Monitoring and Short-Term Variability in Zingst and the Zingster Strom

Monitoring of the Zingster Strom started in 1969 by the State Agency (Wasserwirtschaftsdirektion Küste, now State Agency for Environment, Nature Protection and Geology) on a weekly to monthly basis and includes nutrients, elements and phytoplankton (Schultze and Ventz 1971; Berthold et al. 2018b). Phyto- and zooplankton were investigated more densely near the Zingster Strom since 1969 at least on a monthly basis and later directly in the Zingster Strom (Wasmund and Schiewer 1994; Schumann and Karsten 2006; Feike and Heerkloß 2008) biweekly to weekly. Daily sampling of hydrological parameters and nutrients by the Biological Station started at the Zingster Strom in 1980 (Schumann et al. 2006; Selig et al. 2006). In the following years, more parameters were added.

Chlorophyll a, seston and biological oxygen demand are monitored on a daily basis, total nitrogen and phosphorus on a weekly basis. Since 2000, hydrological parameters (pH, oxygen saturation, conductivity and water temperature) are monitored online every 10 min. Meteorological data including photosynthetic active radiation are obtained every 5 min.

12.4 Salinity, Oxygen and Transparency

The long-term average of *salinity* was 5.6 in the Zingster Strom. Major Baltic Inflows, which happened three times during the last 30 years (Mohrholz et al. 2015), are visible in the salinity data set, but annual salinity means in MBI years are only slightly higher compared to the long-term annual mean values (Schumann et al. 2006). Extremely low salinities were recorded during the summer 2011, when unusually high precipitation led to direct and indirect freshwater inputs. The annual mean of salinity dropped to 4.1 and within the rain period even to a median of 3.4. Salinities above 9 occurred upon flooding and storm events with strong north-west bound winds, which transport Baltic Sea water into the lagoon. However, such events usually last only for some days.

Oxygen saturation was almost always >60% at 08.00 CET (own data from 1996 on, Fig. 12.1), which may be the oxygen saturation minimum time even in summer in well-mixed water bodies after sunrise and before considerable production. The

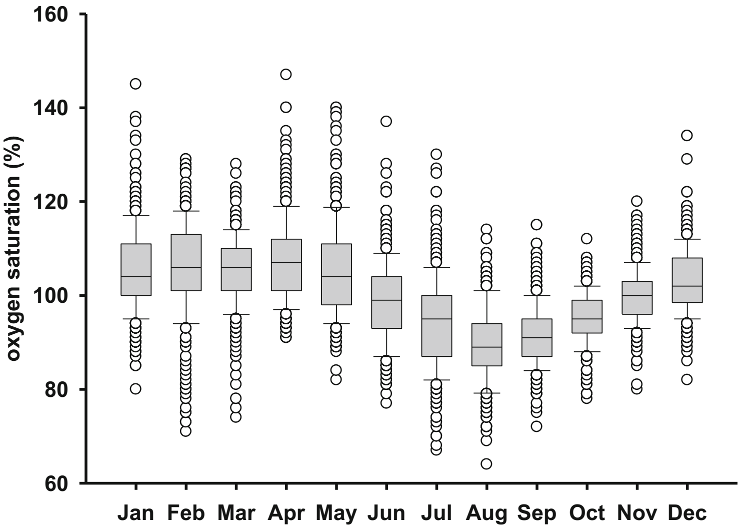

Fig. 12.1 Monthly medians of oxygen saturation (%) at water surface in the Zingster Strom at 8:00 CET: Data set here 2000–2010 (Schumann et al. 2012)

lower values in summer may be the result of a much higher respiration in deeper waters (1–2 m) and sediments compared to the winter season.

Most parts of the shallow lagoon system are well mixed. Own data (unpublished) do not indicate any vertical gradient within the water column. Low-oxygen conditions within the DZBC are restricted to short time periods and a very small part of the lagoon's area such as under stable ice cover (Nasev 1976, Bochert, pers. comm.) and in very shallow water between the reed stands (Karstens et al. 2015). Such suboxic waters are enriched in phosphate due to a high release from sediments (cf. Fig. 12.3 winter of 1995/96 and 1996/97 or Schlungbaum 1982; Karstens et al. 2015).

Underwater light climate, measured as Secchi depth, is constantly low in the DZBC due to high phytoplankton densities (Sagert and Schubert 1999) and was assumed to be further reduced by high sediment resuspension (Meyer et al. 2019; Schumann et al. 2009). The 50 years data set from the Zingster Strom shows a further decrease after 1983. After 2000, higher transparencies are again more often recorded, but mostly in winter. Rarely occurring Major Baltic inflows improve water transparency in the open Baltic and are mirrowed in the Zingster Strom following inflow situations. During the 2003 MBI, highly elevated Secchi depths were measured during 19 days with a maximum of 140 cm (Fig. 12.2).

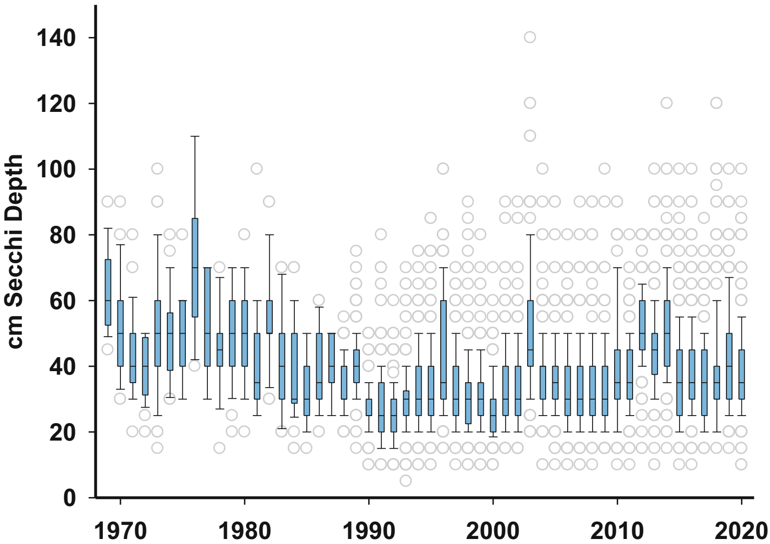

Fig. 12.2 Secchi depth (cm) 1969–2020. 1969–1976 measured weekly to biweekly at two sites within the Zingster Strom, 1977–1887 monthly and from 1987 daily at a central site

12.5 Eutrophication and Nutrients

Two rivers are main point sources of the DZBC, the River Recknitz, mounting into the innermost lagoon, and the River Barthe, draining into the third water basin. Nitrate concentrations in the River Recknitz, the main contributor, vary between 10 and 60 μmol L^{-1} in summer and several 100 μmol L^{-1} compared to ca. 2 μmol L^{-1} (summer) and 8–15 μmol L^{-1} (winter) in the outermost connection to the open Baltic (own unpublished data from 2010). Interestingly, the outermost investigated site (B12/13 near the island Bock) showed about twice as high nitrate concentrations, perhaps influenced by the eutrophied Strelasund region (unpublished data). This indicates that the DZBC, contrarily to many other lagoons of the Southern Baltic, retains most of the incoming nitrogen, thus functioning as an important nutrient buffer towards the open Baltic Sea (Nausch and Nausch 2011).

Nitrate shows a clear *seasonality* with far higher values during winter and early spring. Precipitation washes nitrate into the rivers and directly into the lagoon system. Long-term data from the Zingster Strom show a decrease of these peak values, perhaps due to lower loading from the adjacent terrestrial habitats from land. The slightly increased concentrations in the warmer seasons may indicate internal sources.

The riverine load of phosphate and total phosphorus dropped with the construction of enhanced P-elimination systems in wastewater treatment plants after 1990 (Bachor 2005). In 1985, there was a resolution by the county council for the whole Baltic coastal area concerning an improved manure management (Voigt 1988), which obviously had fast effects on phosphate (Fig. 12.3) and may have at least contributed to the decrease in nitrate peak concentrations after 1988 (cf. Fig. 12.4) by a reduction of diffuse sources.

Since 2003, mean phosphate concentrations in the River Recknitz are as low as 0.6 μmol L^{-1}. Peak values reach 6-8 μmol L^{-1}. Annual medians at the Zingster Strom are only ca. 0.1 μmol L^{-1}, which is just above the determination limit (Schumann et al. 2012). Phosphate has no seasonality, but can be released from the sediments, if oxygen concentrations drop (see above). Though P release from sediments is potentially high (Bitschowsky 2016), we assume an overall low internal P loading in the DZBC, as this release is a rare and/or locally very restricted event. Such P release from sediments has been observed locally in reed belt puddles on warm days (Berthold et al. 2018a; Karstens et al. 2015), or in the rather rare situation of harsh winters with a longer ice cover. In both ice-winters of 1995–1996 and 1996–1997, short periods with drastic increases of PO_4^{3-} were observed in the Zingster Strom, accompanied by a “foul” smell of H_2S (early 1995) and a drop in measured oxygen saturation (January of 1997) (Fig. 12.5). In contrast, phytoplankton can take up any phosphate pulses extremely fast and even mobilise it enzymatically (Berthold and Schumann 2020).

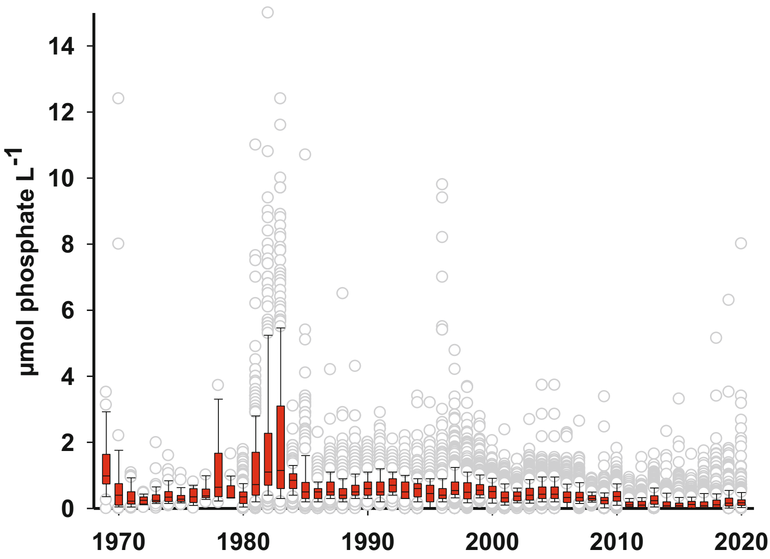

Fig. 12.3 Phosphate concentrations (μmol L^{-1}) in the Zingster Strom as annual median based on biweekly to monthly values until 1980, after that daily measurements. Box (red)-whisker (black)-plots; outliers are indicated as grey circles

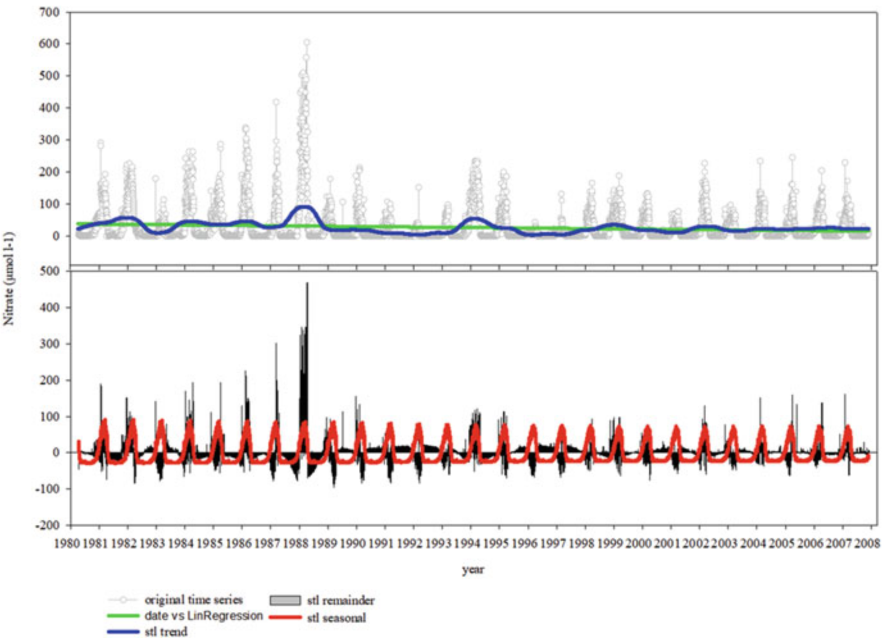

Fig. 12.4 Nitrate concentration (μmol L^{-1}) in the Zingster Strom based on daily measurements. Grey: original data, green: overall linear regression line, blue: long-term non-linear trend (a)

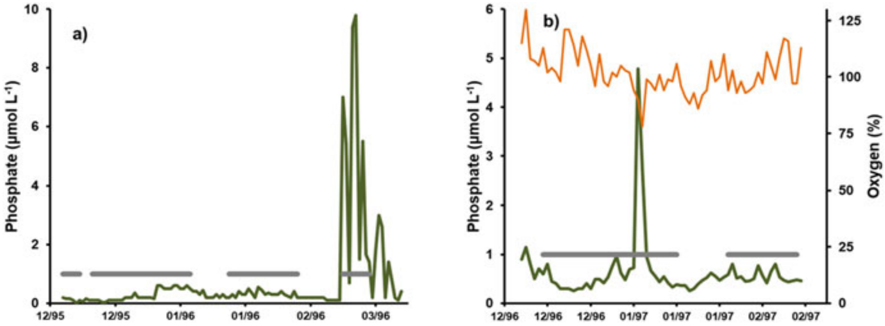

Fig. 12.5 Phosphate concentrations (µmol L^{-1}) in the Zingster Strom during the ice winters of 1995–1996 (**a**) and 1996–1997 (**b**), respectively. Oxygen saturation was 0.7–15.1% within the shallow water of five investigated sites along the salinity gradient on 04 March 1996 and is only continuously available for the second winter at the water surface. Ice cover periods are illustrated with grey bars

12.6 Phytoplankton Biomass and Composition

Species numbers of microalgae amounted to a total of 429 over many years and sites within the DZBC (Kell et al. 1975). Species composition of green algae seems to be stable, but absolute biomass dropped from the mid 1990s on (own data). Spring blooms of diatoms became rare after 1991 (Wasmund and Börner 1992). This loss has probably a major impact on the trophic efficiency in the DZBC (see Chap. 13). Long-term data show an increase of the share of cyanobacteria in phytoplankton as well as a shift in cyanobacteria composition. Seven cyanobacterial genera of high dominance in the 1970s and 80s became rare and three morphotypes increased their biomass. All of the latter belong to colonies forming alpha-picocyanobacteria (Synechococcus-Prochlorococcus-Cyanobium-clade), (Albrecht et al. 2017). In Reynolds et al. (2002) they were addressed as *Microcystis reinboldii*, *Aphanothece* spp. and *Coelosphaerium* spp. of the Chroococcales. Gessner (1938) described already such “*Microcystis*” and “*Coelosphaerium*” colonies. Single cells were most likely overlooked until 1990. Also many filamentous cyanobacteria became more important. They belong to the clade *Pseudanabaena* within the group of Synechococcales, some of them peak in autumn (Fig. 12.6).

Total phytoplankton biovolume increased until about 2005, after which a decrease seems to have occurred (Fig. 12.7). This shift is, however, not accompanied by a reduction in Secchi depths (*c.f.* Fig. 12.2). A possible explanation is that seston constitutes only to a small portion of phytoplankton (Schumann et al. 2001), so that a phytoplankton reduction may only have a minor impact on underwater light climate.

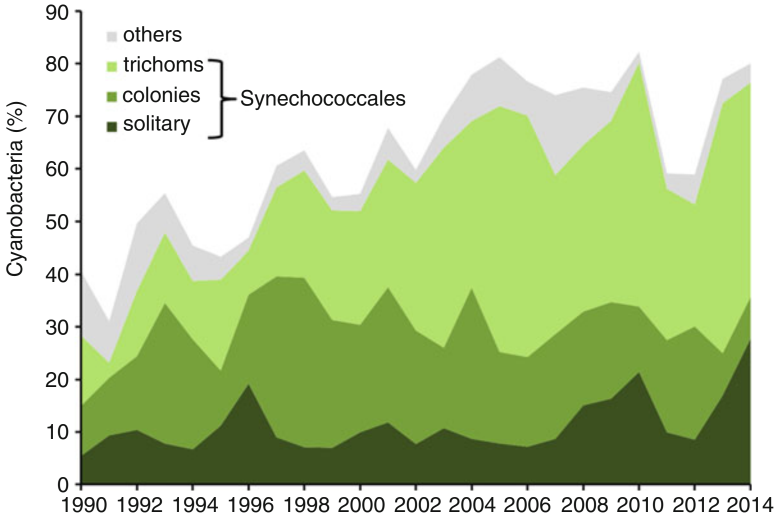

Fig. 12.6 Composition of cyanobacteria in the phytoplankton of Zingster Strom. From Schumann et al. (2019), modified

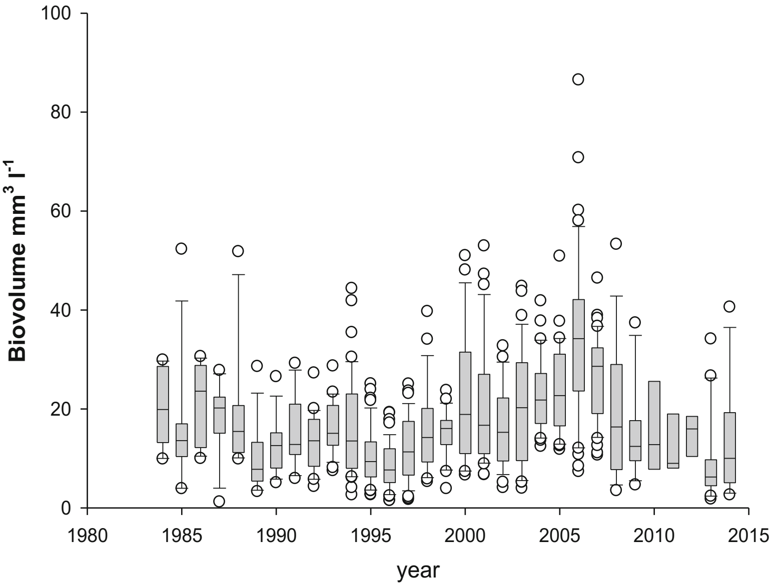

Fig. 12.7 Long-term data of phytoplankton biovolume ($mm^3\ L^{-1} \approx$ mg fresh mass L^{-1}) in the Zingster Strom: lines in the boxes are the median values of about 30–40 samples per year since 1991, grey boxes incorporate 50% of all average values, whiskers are the 10 and 90% percentiles, circles are outliers

12.7 Primary Production

Though an important ecosystem function, primary production is only rarely measured in sufficient seasonal and spatial resolution. Moreover, the results are strongly dependent on methods, data evaluation and conversion factors. This is the case for all compartments of primary production such as submerged macrophytes, epiphyton and phytoplankton.

pH is a frequently measured parameter and can serve as a proxy for (total) primary production and can be used as such in the DZBC, which shows a distinct increase of pH during summer (Fig. 12.8). For the DZBC, values as high as 9.40 and even 9.92 (Nasev 1976; Börner 1984) have been reported. Since 1990, mean pH has dropped though high values still occasionally occur (Fig. 12.8). In shallow bights with dense macrophyte vegetation, pH can still reach values of ca. 10 (own observation).

12.8 Discussion

For the DZBC, only episodic observations are available for the first half of the 1900s century, when eutrophication caused major changes in physico-chemical as well as biological parameters such as a massive decline of submerged macrophytes (Gessner 1938; Lindner 1972, 1976, 1978; Behrens 1980, 1982; Teubner 1989). Denser observations are available since 1970 and have continuously been improved. Today, a long-term data set with high-resolution is available.

The results show the importance of this data set. Only in a long-term data set, the impact of extreme, but rarely occurring events can be identified and quantified. The data set shows an increase of water column nitrate concentrations and decrease of salinities during periods of elevated precipitation, drastic increases of salinity and

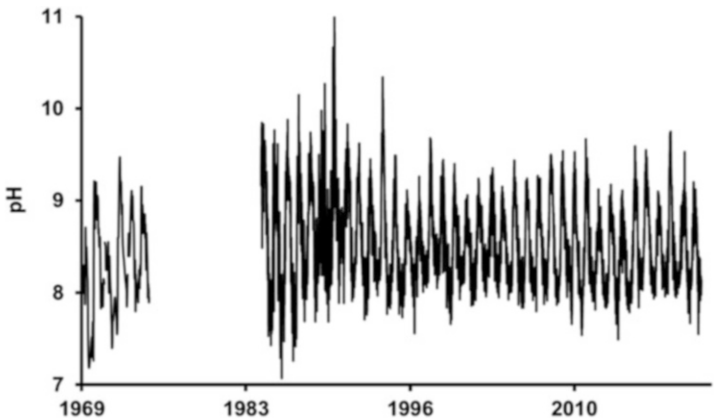

Fig. 12.8 Long-term data of pH in the Zingster Strom, 1969–1976 weekly-monthly, from mid 1984s on daily in the Zingster Strom

water transparency during Major Baltic Inflows as well as severe storm events causing inflows of water from the open Baltic into the lagoon system, and short-term drastic increases of P release from the sediments into the water cover during ice-winters (see above). Similar drastic changes are expected also to happen in other coastal lagoons such as the Vitter Bodden, but cannot be identified as high-resolution long-term data are not available from in this ecosystem.

The combination of these data with intensive investigation campaigns such as the BACOSA project, and with (equally highly resolved) weather data allows for a detailed analysis of both food-web changes (see Chap. 13) and the impact of climate change on water quality of and, finally, ESS provided by the DZBC. Periods of elevated precipitation are predicted especially for the winter and may cause an increase of water column nutrients (especially nitrate), while lower frequencies of ice-winters may reduce the risk for P release from the sediments and thus internal eutrophication.

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Carbon Fluxes/Food-Webs: Effect of Macrophytes on Food Web Characteristics in Coastal Lagoons

13

Martin Paar, Maximilian Berthold, Rhena Schumann, and Irmgard Blindow

Abstract

At the land–sea interface, coastal waters and especially lagoons are prone to multiple anthropogenic pressures impacting ecosystem functioning. In the Baltic Sea, eutrophication, defined as increased nitrogen discharge in coastal waters, is one of the main ecosystem threats. A change in the ratios between the different nutrient, e.g. nitrogen and phosphate, influences the communities of primary producers at the base of the food web, as different types of primary producers (e.g. macrophytes and phytoplankton) have specific nutrient requirement. The type of primary producers influences the composition and availability of food resources for consumers, and how primary production enters and passes through the food web. Thus, ecosystem functioning of lagoons dominated by, e.g. phytoplankton will differ from that of lagoons dominated by, e.g. macrophytes. Commonly, higher nutrient discharge sustains higher ecosystem production. However, in lakes, it has been shown that high eutrophic systems had lower total production than non-eutrophic ones, a concept known as “the paradox of enrichment”. Here, we used ecological network analyses on carbon flow networks to compare the functioning and structure of two lagoon ecosystems in the German Baltic Sea under different eutrophication pressure: the highly eutrophic Grabow estuarine lagoon in the Darß-Zingst Bodden chain dominated by phytoplankton, and the less eutrophic Vitter Bodden marine lagoon,

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dominated by macrophytes. The Grabow lagoon, dominated by phytoplankton, had higher redundancy in longer trophic pathways and higher recycling rate than the macrophyte-dominated Vitter Bodden lagoon, characterised by a more specialised food web. The overall food web production was higher in the less eutrophic Vitter Bodden lagoon than in the highly eutrophic Grabow lagoon. Our results, showing a lower ecosystem production with an increased nutrient discharge, confirms empirically and for the first the "paradox of enrichment" in coastal waters.

13.1 Introduction

Carbon cycling in aquatic ecosystems is complex. On its way through the food web, carbon is transformed between inorganic and organic, dissolved and particular, dead and living forms. Knowledge of this carbon cycling is essential to understand ecosystem structure and functioning, including its behaviour under different anthropogenic pressures. The effect of eutrophication on the carbon cycling has been investigated in detail within the Darß-Zingst Bodden chain (Schiewer 1998). Eutrophication was shown to alter carbon cycling through shifting primary production to the pelagic realm, enhancing microbial activity and increasing the amount of unused primary production in the system (Schiewer 1994). Within the BACOSA project, we extended the existing data on carbon cycling in the pelagic realm with data on macrozoobenthos and fish fauna and integrated higher trophic levels in the analysis of carbon cycling. Thereby, we compared two lagoons under different eutrophication pressure, i.e. the estuarine lagoon Grabow (GB) in the Darß-Zingst Bodden chain (DZBC) and the marine lagoon Vitter Bodden (VB) in the Westrügensche Bodden chain by means of ecological network analysis (ENA). ENA has already been used to analyse food web behaviour under anthropogenic pressures in other shallow coastal aquatic ecosystems (Schückel et al. 2015; Paar et al. 2019). The methodology consists of a set of algorithms to analyse the flows of carbon, nutrients or energy between all compartments of a food web. ENA reveals ecosystem properties that are otherwise not evident from direct observation (Fath et al. 2007) and can thus be used to gain an integrative, holistic and unbiased assessment of aquatic ecosystems (de la Vega et al. 2018; Safi et al. 2019). The analysis was based on an intensive investigation in both lagoons at around 1 m water depth during 2017 (Paar et al. 2021; unpublished data). Combined with other data obtained during the BACOSA project, we finally use this analysis to put both lagoons into the context of the "paradox of enrichment" and the "alternative stable states" hypothesis.

13.2 Primary Producers

In the VB, macrophyte densities increase with water depth and reach up to 70% cover from 1 m to the maximum depth of 2.8 m. Down to 1 m depth, *Ruppia* spp. dominates the vegetation with patches of *Chara* spp. and *Fucus vesiculosus*. Between 1 and 2 m depth, *Stuckenia pectinata* is dominating. *Zostera marina* grows below 1.5 m depth and dominates below 2 m (Blindow et al. 2016; Bühler 2016).

Primary producer biomasses in the VB were dominated by benthic life forms, such as macrophytes (vascular plants and macroalgae) and filamentous epiphytic algae, while phytoplankton biomasses were negligible. The net primary production was dominated by the benthic compartment with a daily average of 2.7 g C m^{-2} d^{-1} between March and October. Epiphytic filamentous algae were the main contributors to net primary production in 2017 (Paar et al. 2021). Pelagic net primary production only reached half of the intensity of the macrophytic net primary production and was highest in early summer, but pelagic productivity (i.e. production per unit biomass) was higher than the benthic primary productivity (Fig. 13.1). Estimated annual values of benthic net primary production of 937 g C m^{-2} y^{-1} were comparable to other coastal and transitional waters covered with macrophytobenthos (Duarte et al. 2005).

The macrophytes in the GB, which were dominated by *Stuckenia pectinata*, reached a maximum depth of only 1.5–2.0 m and formed a sparse vegetation

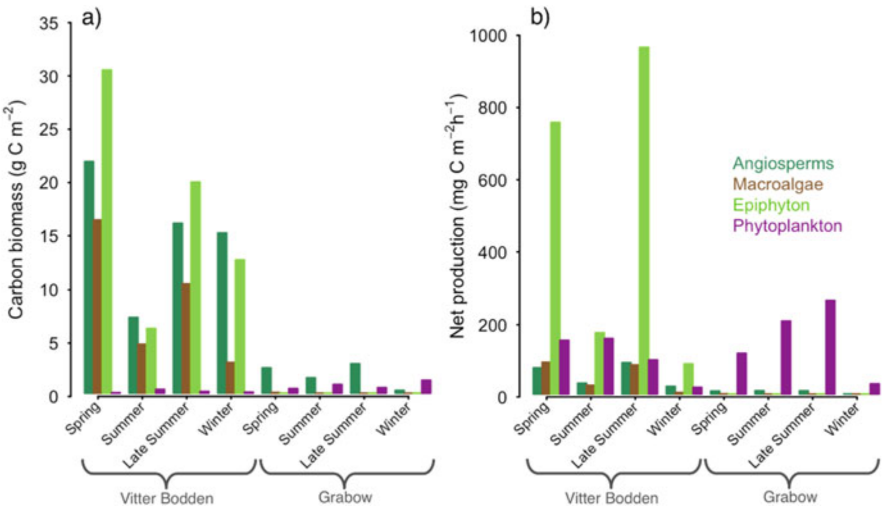

Fig. 13.1 Average biomass (**a**) and daily net production (**b**) of benthic and pelagic primary producers in Vitter Bodden and Grabow during 2017. Phytoplankton production was experimentally determined by oxygen evolution in light and dark bottles. The benthic production from angiosperms, macroalgae and epiphyton was calculated using species-specific photosynthesis parameters from the literature combined with own biomass data

cover, but still represented the major share of total primary producer biomass in summer at the sampling site. Phytoplankton contributed to only about 30% of total primary producer biomass in summer, but on average to 98% of total areal net primary production, while the macrophytes only represented 2% (Paar et al. 2021). While phytoplankton biomasses in the GB were around one order of magnitude higher than in the VB, net primary production of phytoplankton was similar in both lagoons (Fig. 13.1). Low phytoplankton productivity is confirmed by other investigations of the DZBC showing high water turbidity as a consequence of eutrophication. High phytoplankton biomass increases water turbidity and deteriorates the underwater light climate. It can cause negative depth-integrated net primary production, i.e. respiration outweighs the photosynthesis already in relatively shallow water (Schumann et al. 2005).

13.3 Consumers and Carbon Fluxes

The vegetation of the VB is characterised by high spatial complexity, which may allow the coexistence of several filtrating and grazing species. Throughout the year, the macrozoobenthos was dominated by filter-feeding bivalves, mainly *Cerastoderma glaucum*, *Mytilus edulis* and *Mya arenaria*, which contributed to 55% (annual average) of the macrozoobenthic biomass, while grazers, dominated by Hydrobiidae, *Gammarus* sp. and *Idothea chelipes*, contributed to 21% (Fig. 13.2).

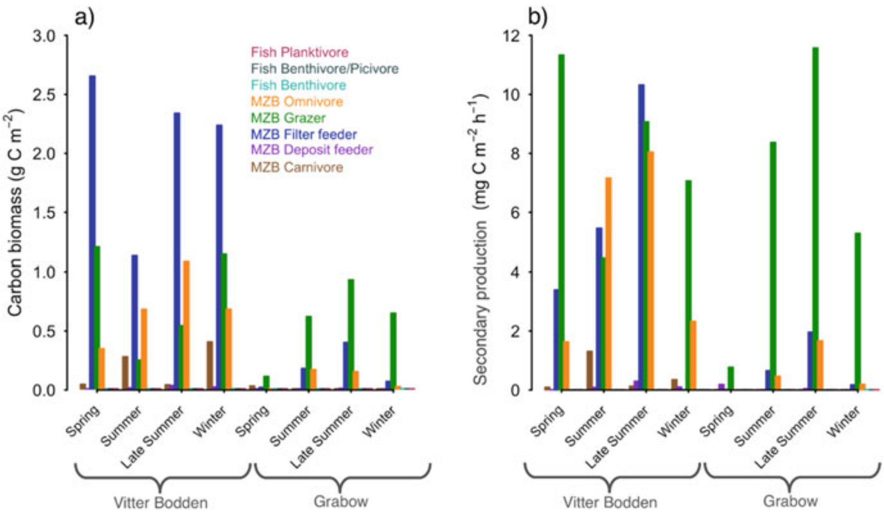

Fig. 13.2 Average biomass (**a**) and daily net production (**b**) of macrozoobenthos and fish sampled in Vitter Bodden and Grabow during the entire season 2017. Macrozoobenthos and fish were grouped in feeding guilds

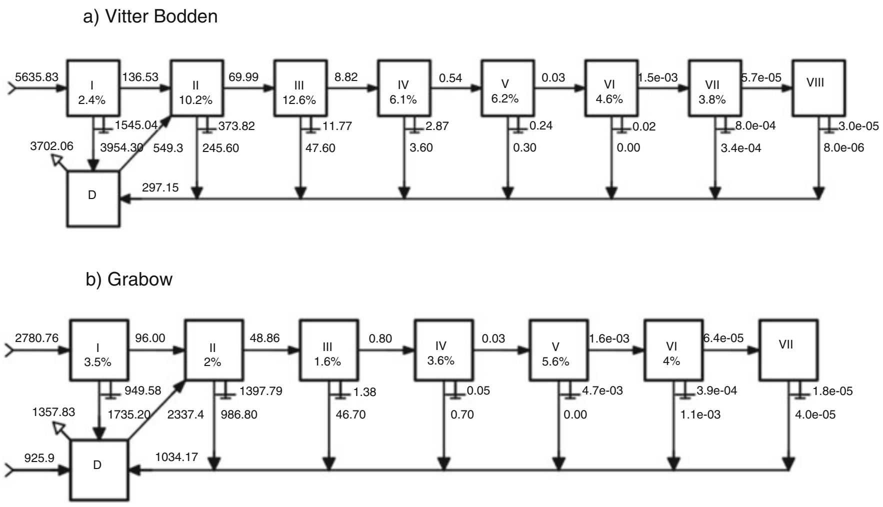

Fig. 13.3 Lindeman spines for (**a**) Vitter Booden and (**b**) Grabow in summer 2017. The percent value in each of the trophic level indicates the ratio between carbon of input into that level and the amount passed to the next higher trophic level. Each box represents a discrete trophic level. Flows are shown for exogenous inputs, exchange between trophic levels, and losses to the surrounding environment (exports and respiration) from each trophic level in mg C m^{-2} d^{-1}. *D* detritus pool

In the VB, consumers represented 5% of the total biomass and 7% of the total production of the system. Consumers, including filter feeders, grazed 17% of net primary production in summer. Bacteria had a carbon turnover rate twice as high as macrozoobenthos. Zooplankton, including protozoans, rotifers and copepods, grazed 2% of phytoplankton net primary production, but reached temporarily far higher values (Blaffert 2018). The production of filter-feeding bivalves depended on average to 61% on the pelagic primary production and to 30% on suspended detritus in summer. Grazers consumed mainly macrophytes and their epiphytes. Crucially, only 10% of the proliferating, epiphytic filamentous production was entering the food web directly or indirectly. Fish consumed on average 4% of the macrozoobenthic secondary production. Dominant Gobiidae were relying through their macrozoobenthic diet on carbon originating from epiphytic production, sediment organic matter and meiofauna in summer. Gobiidae were the most important food source for perch (*Perca fluviatilis* L.), which had much higher growth rates in VB than in GB (Rittweg 2019). Secondary production was transmitted to higher trophic levels with an average trophic efficiency of 5% in the food web. Eight trophic levels were present in the food web, and perch and flounder (*Platichthys flesus* L.) were both occupying the highest trophic level (Fig. 13.3).

In the GB, species numbers, biomasses and production of consumers were lower than in the VB (Fig. 13.2). Some filter feeders (*Mytilus edulis*, *Parvicardium hauniense)* and the grazer *Theodoxus fluviatilis* were absent, since the substrate and refuge formed by the macrophytes were too sparse. Grazers, mostly

Hydrobiidae, represented 74% (annual average) of the macrozoobenthic biomass, while filter feeders, dominated by *Cerastodermum glaucum*, only contributed 12%.

Total carbon cycling (total system throughput) of GB was 20% lower than in the VB, and 4% of the net primary production was entering the food web through direct grazing. Bacteria represented 5% of the total biomass and 23% of the total system production. Pelagic and sediment bacteria expressed a ten-fold higher activity than all other consumers in summer and consumed indirectly 34% of phytoplankton net primary production. However, bacterivory was 12% lower than in VB. The meiobenthos fed 4% of the production of sediment bacteria. Less than 1% of the pelagic bacterial production was consumed. Detritus had similar contribution as primary production to consumers' diet. Zooplankton directly grazed 2% of the pelagic primary production and 3% of the suspended particulate material in the water column. Pelagic ciliates were strongly dependent on bacteria. Macrozoobenthic grazers were depending mostly on microphytobenthos consuming 18% of its production, while filter feeders consumed 0.1% of the pelagic primary production. Only few macrozoobenthic species showed a high biomass and abundance, which further reduced the prey availability for higher trophic levels. Fish represented less than 0.1% of the total system production and consumed 2% of the macrozoobenthic production. As a result, the average trophic transfer efficiency was low with only 3% (Fig. 13.3).

13.4 Differences of Food Web Characteristics Between Both Lagoons

The size of the system (total system throughput) was significantly larger in VB (Fig. 13.4). Here, macrophytic cover was high during summer, but most of the macrophytic production was not consumed reflected by the higher dissipation of energy (overhead), lower recycling of material and shorter cycles within the food web than in GB. Lower redundancy of the food web indicated a more specialised diet of the primary consumers each feeding on a specific type of primary production, with grazing being equally distributed among primary producers. Low trophic depth shows a system that is mostly dependent on short pathways of carbon cycling. The higher complexity (effective link density) of the system with macrophytes could be explained by the integration of several benthic and pelagic grazing chains by higher trophic levels.

High macrophyte species number corresponded to high variability in number of growth strategies and thus, increased length of the macrophytic growth period. As shown for other vegetated habitats, high species number of macrophytes may stabilise macrophytic net primary production on a high level throughout the season (Middelboe and Blinzer 2004). The large seasonal variation of the macrophytic C:N ratios in the VB indicates that macrophytes differ in their attractiveness as food. Also consumer composition and biomass varied, suggesting different grazing pressures and considerable differences in food web structure and functioning among seasons.

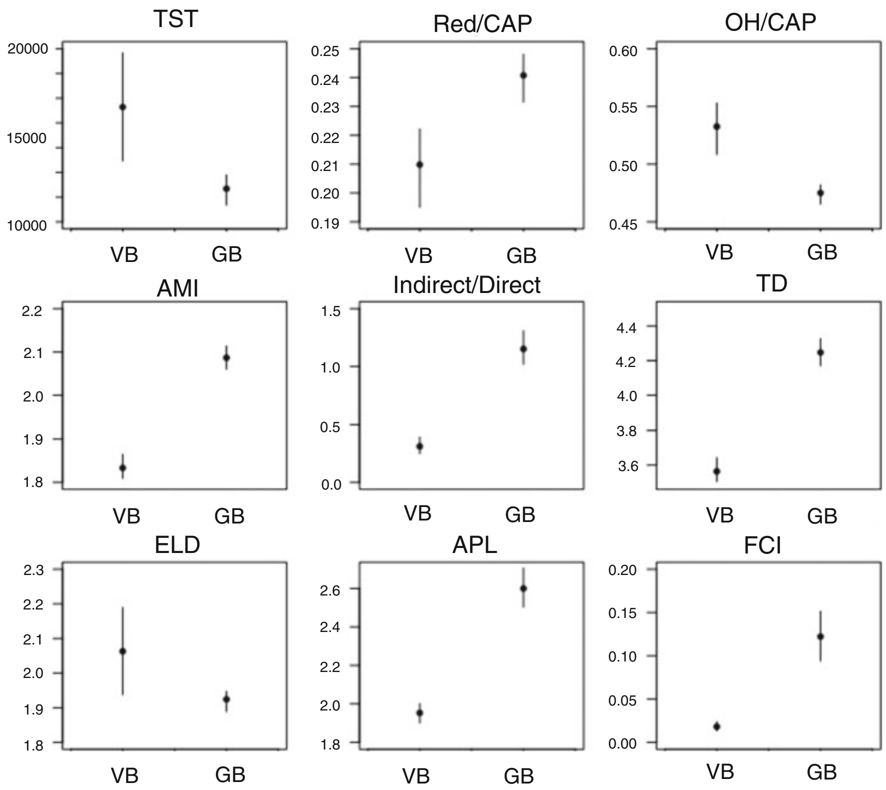

Fig. 13.4 Whole system indices for Vitter Bodden (VB) and Grabow (GW) in summer 2017. The point represents the initial food web parameterisation and the line above and below shows ±95% confidence interval (Con Int 95%) based on 10,000 plausible food web parameterisations revealed by uncertainty analysis. A significant difference is assumed between the two lagoons system indices when the ConInt 95% does not overlap. *Tst* total system throughput, *Red/CAP* relative redundancy, *OH/CAP* relative overhead, *AMI* average mutual information, *Indirect/Direct* ratios of indirect over direct flows, *TD* trophic depth, *ELD* effective link density, *APL* average path length, *FCI* Finn cycling index calculated with the total system throughflow

In the more eutrophicated lagoon GB, bacterial activity was closely coupled to phytoplankton production resulting in low energy loss (overhead), long cycles and high recycling of carbon in the food web (Fig. 13.4). The dominance of bacteria was reflected by highly constraint carbon flows and overall low connectivity of the food web. Primary production was shunted through the microbial loop and became inaccessible to higher trophic levels, explaining low accumulation of biomass, secondary production and trophic efficiency of consumers. The system was relying on external import to support microbial activity. An equal ratio of detritivory to herbivory in combination with high recycling suggested carbon was mainly exchanged over two or more compartments connecting adjacent parts of the food web and making the system prone to indirect effects. Macrozoobenthic consumers relying mainly on the same food source explained the high redundancy in carbon flows.

The dominance of bacteria and small consumers in the pelagial and benthos accelerated turnover processes, which may enable the system to react quickly to external drivers. High recycling and dominance of indirect flows indicated that internal processes such as competition and cascading effects mainly influence the flow of carbon. Without efficient grazing control, the high phytoplankton biomass reduces light availability and thus, suppresses the recovery of habitat-forming macrophytobenthos (Paar et al. 2021) and keeps the ecosystem in a turbid, phytoplankton-dominated state (see below, alternative stable states; Chap. 12).

13.5 The "Paradox of Enrichment"

It may seem out of question that higher nutrient supply causes higher ecosystem production. By means of theoretical calculations combined with laboratory experiments, however, already Rosenzweig (1971) showed that nutrient enrichment increases population oscillations and system instability, which finally results in reduced production. This "paradox of enrichment" has also been observed in natural ecosystems (e.g. Davis et al. 2010), but seems to be the exception rather than the rule. This has been explained by the fact that ecosystems rarely reach any climax of succession due to disturbances, that their food webs are far more complex than laboratory systems, and that prey organisms had quite an amount of time to "learn" to avoid their predators (Roy and Chattopadhyay 2007).

Aquatic ecology textbooks (see Table 8.4 in Schwörbel and Brendelberger 2013) give higher general values for daily net primary production in mesotrophic than in eutrophic lakes. Several investigations show, however, that in shallow lakes, nutrient enrichment can cause reduced production already at the level of primary producers. This "paradox of enrichment II" has been explained by high light limitation and self-shading in highly eutrophic ecosystems (López-Archilla et al. 2014; Blindow et al. 2006).

Results obtained during the BACOSA project confirm these results from freshwater lakes. System net primary production was lower in the highly eutrophic GB than in the less nutrient-loaded VB (see above), which can be explained by a high degree of self-shading (Schumann et al. 2005). Also for higher trophic levels, our results show lower production in the more nutrient-rich system. While benthic primary production in GB is suppressed by low light availability, the phytoplankton resists grazing by excretion of mucoid substances and/or inadequate food quality (Schumann et al. 2001) and thus causes low system trophic efficiency. Reduced refuge availability through the loss of submerged vegetation additionally may restrict zooplankton and thus grazing pressure on the dominant phytoplankton (Blaffert 2018; Schumann et al. 2009). Our results confirm Schiewer (1998) who described a decrease of trophic transfer rates during transition from a macrophyte-dominated clearwater state to a phytoplankton-dominated turbid state in the DZBC, caused by increasing eutrophication. Our ENA analysis shows that unused carbon is not accumulated, as suggested by Schiewer (1998), as the size of the system was significantly lower in the eutrophicated lagoon. The BACOSA project gives the first empirical support for the "paradox of enrichment" hypothesis in brackish ecosystems. A similar "paradox of enrichment" phenomenon is assumed to occur in a number of coastal lagoons around the Baltic Sea, as these highly eutrophic ecosystems (see Chap. 18) are exposed to increasing light limitation of photosynthesis, if nutrient loadings are further enhanced. In contrast, the less eutrophic outer coastal waters and the open Baltic Sea show increasing primary production with progressive eutrophication (see Chap. 17). Decline of macrophytes diversity and biomass during eutrophication causes lower production of higher trophic levels including organisms with commercial interest, which suggests a negative impact on ecosystem services.

13.6 Alternative Stable States

Though they are situated in the same geographical region and hydrologically connected to each other, the ecosystem structures and functioning of the two studied lagoons were significantly different (Fig. 13.4). Apart from different hydrological regimes, these differences can be explained by the difference in dominant primary producers and how carbon is entering the food web and sustains its production.

Coastal Baltic lagoons have been suggested to occur in different alternative stable states (Dahlgren and Kautsky 2004; Rosqvist et al. 2010; Austin et al. 2017), which have been described for a number of shallow aquatic ecosystems, with dominance of macrophytes at low and phytoplankton at high nutrient concentrations. In an intermediate nutrient range, both macrophyte or phytoplankton dominance is possible, which is why such ecosystems are assumed to occur in alternative stable states. At progressive eutrophication, submerged vegetation counteracts the increasing water column turbidity by a number of habitat-stabilising mechanisms such as nutrient accumulation, reduction of sediment resuspension and a refuge for zooplankton. Once a critical turbidity is reached, the system switches rapidly to a phytoplankton-dominated "turbid" state, accompanied by a drastic increase of turbidity due to the breakdown of the feedback mechanisms. A return to the "clearwater" state with abundant submerged macrophytes is only possible if nutrient loadings are reduced far below the point where the switch happened before (Scheffer et al. 1993). These ecosystems are thus characterised by a high resilience and non-linear response to external forces: The vegetation is able to counteract increasing nutrient loadings for a long time. Close to the "tipping point", small external impacts may cause a drastic change of the whole ecosystem, with major consequences for ecosystem services (see also Chap. 28).

In the VB, submerged vegetation still forms dense vegetation down to the deepest parts (Bühler 2016). Since the 1930s, no major change in density and distribution of this vegetation seems to have occurred, but a shift in species composition from low "bottom-dwellers" to taller "canopy-formers" has been observed. Bottom-dwellers are favoured by low nutrient concentrations and a high frequency of ice-winters, why this shift has been interpreted as an “early warning signal” for both eutrophication and climate change (Blindow et al. 2016).

Also the results obtained during the intensive investigation in 2017 indicate reduced system stability. The high share of filamentous, epiphytic algae in total vegetation biomass may shorten the macrophytobenthic growth period, cause the loss of biodiversity and reduce habitat stability (Paar et al. 2021). Trophic feedback mechanisms including temporarily high grazing pressure from zooplankton (Blaffert 2018; Meyer et al. 2019) and grazing control of epiphytic algae normally stabilise the submerged vegetation by improving the underwater light climate. In 2017, proliferating epiphytic algae may have outgrown grazing control of the food web weakening trophic feedback mechanisms and causing increased dissipation of energy and reduced organisation of the food web. Despite high connectivity of the food web, the VB is characterised by low recycling, redundancy and short trophic pathways. This gives rise to the assumption that this system might be close to the “tipping point”. Such a transition between the clear and turbid state is called the “crashing” state and was described for freshwater (Sayer et al. 2010; Verhofstad et al. 2017) and seagrass ecosystems (Viaroli et al. 2008).

In the Darß-Zingst Bodden chain as in other coastal lagoons of the Baltic Sea, submerged vegetation has collapsed during the 1980s after a longer period of high nutrient loadings. The lagoon is still dominated today by high phytoplankton densities in spite of a considerable reduction of nutrient loadings (Schiewer 1998; Munkes 2005; see Chap. 12). The sparse submerged vegetation in the GB is associated with a reduced system size and activity with an overall reduced trophic efficiency, indicating compromised trophic control mechanisms within the system. Contrary to our expectations, the eutrophicated GB is characterised by significant higher values of redundancy, recycling and organisation of the food web indicating a more stable system than the VB. For decennia, this system has been dominated by small, grazing-resistant and nutrient-efficient cyanobacteria, which furthermore are highly adapted to the shifting light conditions of the predominant Langmuir situation and thus superior to other pelagic primary producers, especially in a situation of high phytoplankton densities (Scheffer et al. 1994; see Chap. 12). Once they have reached dominance, these cyanobacteria are suggested to cause persistence and high resilience of the turbid state.

While the VB is assumed to occur in a rather instable clearwater state with high production in all trophic levels, the GB seems to be in a stable turbid state with low overall production.

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14 Ecological Structure in Benthic Habitats of Offshore Waters

Mayya Gogina and Michael L. Zettler

Abstract

Mapping the structure of benthic macrofauna communities is essential for assessing and monitoring the state of the seafloor habitats. The presence and dominance of species and functional traits they exhibit alter the biotic and abiotic settings and provide a variety of ecosystem services. Species distribution influence biogeochemical fluxes, transport and food webs. Based on the aggregated data, benthic faunal communities and their traits in the offshore waters are mapped. Key players and hot spots of functional aspects are identified, supporting the projections of marine ecosystem features. We discuss the sources of variability and compare bioturbation values measured directly with those captured by corresponding indices, in an attempt to highlight this functional aspect of benthos.

The spread of species, the arrangement of their life histories and functions they perform are tied to their physical environment. Within soft-sediment communities, tube-building polychaetes attenuate flow and stabilise the sediment (Jones and Jago 1993), bioirrigation behaviour alleviates biogeochemical variation. Filter feeders, like mussels, remove suspended matter from the water column and often dominate in sandy habitats. Deposit feeders, like polychaetes, burrow and sift sediments for detritus and may prevail in fine-grained mud (Anderson et al. 2010).

Distinct benthic macrofauna communities (sensu Avellan et al. 2013), with specific ecological functions, often go beyond the borders of biotopes pre-defined by specific classification systems (see Chap. 8). In the German EEZ species richness is relatively high compared to the entire Baltic Sea, and communities are

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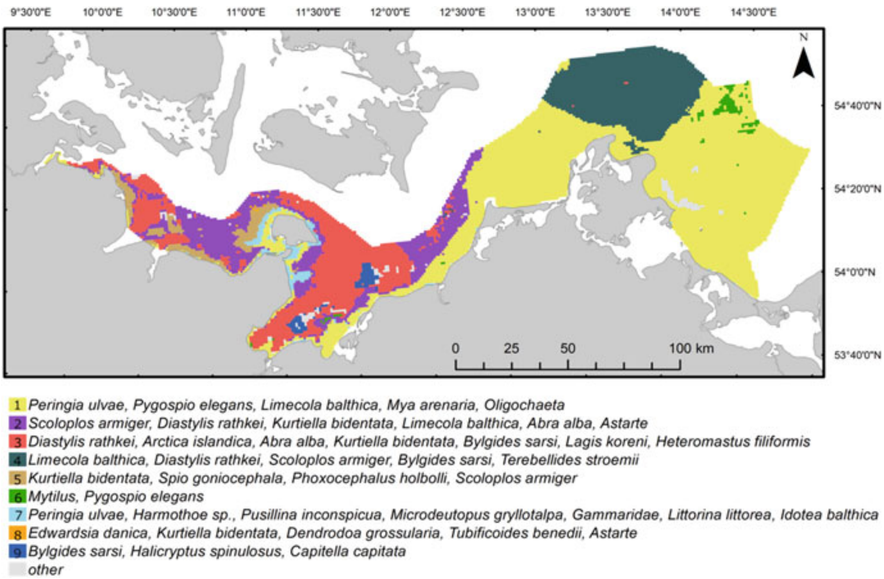

Fig. 14.1 Full-coverage map of abundance-based communities' distribution predicted by Random Forest based on abiotic layers described in Gogina et al. (2020). Note: overall OOB estimate of error rate is 16.9%, and accuracy is particularly low (33%) for community 6. Predicted coverage area per community is: 1—6616 km^2, 2—1631 km^2, 3—2097 km^2, 4—2113 km^2, 5—382 km^2, 6—140 km^2, 7—244 km^2, 8—8 km^2, 9—122 km^2

characterised by more than one or few dominant species. This biological diversity should be considered to examine the variation of communities.

Only the records from the seafloor habitats of the German Baltic Sea part reflect this immense diversity (Zettler et al. 2018), containing over 420 species in 14 phyla, including:

129 species of annelids (aquatic worms), 120 species of arthropods (crabs, shrimp, barnacles), 89 species of molluscs (clams, sea slugs, snails), 33 species of cnidarians (hydrozoans, anemones, jellyfish), 33 species of bryozoans (crusts, bryozoans), 7 species of echinoderms (sea stars, sea urchins, sea cucumbers), 7 species of poriferans (sponges), 6 species of chordates (sea squirts), 6 species of nemerteans (ribbon worms), 2 priapulids (penis worms), 2 entoprocts, 1 phoronid, 1 species of platyhelminths and 1 species of sipunculas (peanut worms).

Being linked across scales, various characteristic communities and species will be derived based on different spatial and temporal resolution and extent. For general community mapping data often are aggregated at the scale of decades to attain sufficient spatial coverage. We used data from 2000 to 2018 and delineated benthic faunal communities in the German part of the southern Baltic based on abundance (Fig. 14.1), using the methods described in Gogina et al. (2016). Salinity and its variability, substrate, oxygen conditions and depth (as a cumulative predictor of multiple factors including food availability and light penetration) determine the distribution of species and structure of communities in the southern Baltic.

In total we distinguished nine macrobenthic communities (Fig. 14.1). Variations in traits of inhabiting communities are reflected in their influence on sediment ecosystem functioning. Biomass is often better related to functional aspects compared to abundance. For example, the filter-feeding capacity of an individual of large bivalve specimen (e.g. *Arctica islandica* or *Mya arenaria*) would greatly exceed that of many small polychaetes (e.g. *Pygospio elegans*). In turn, for *Mya arenaria* highest filtering capacity in the study area was estimated in the southern Pomeranian Bay, close to the mouth of Oder River (Darr 2015). On the other hand, we present a spatial trend from communities dominated by long-lived and highly specialised species towards those with prevailing short-lived ubiquitous species along a gradient of decreasing salinity from west to east. Observed dominance of ubiquitous species and increase of functional redundancy towards the brackish waters in the east of the study area seems to buffer the functional loss, thereby increasing the robustness of the benthic ecosystem to environmental changes (Darr et al. 2014).

Bioturbation is another important function contributing to higher-level ecosystem processes such as nutrients recycling, bentho-pelagic coupling, and burial or release of contaminants. Bioturbation potential (BPc, Solan et al. 2004) is a quantitative trait-based indicator for ecosystem functioning that combines particular traits (e.g. sediment reworking and mobility) with species abundance and biomass. This combined term infers a coarse indication of bioturbation capacity, showing an only moderate relation to tracer-based measurements of observed bioturbation patterns. Bioturbation potential has an advantage as a suitable indicator for obtaining full-coverage maps. Structural differences (composition, density, biomass and diversity) in communities are reflected in different magnitudes of the estimated bioturbation potential with hotspots predicted for the Pomeranian Bay, in the south-western part of the marine trench “Kadetrinne”, as well as deeper parts of the Kiel Bay west of Fehmarnbelt (Fig. 14.2). *Hediste diversicolor* (with highest average densities associated with community 1, see Fig. 14.1) had the overall highest contributions to total BPc in mud, medium and coarse sands (with respective contributions declining from 23 to 15%). In fine sands *Arctica islandica* and *Peringia ulvae* had the highest contribution (Gogina et al. 2020).

Over the considered scale (i.e. spatial, from Flensburg Fjord to Swinoujscie, and temporal, from 2000 to 2018), spatial variability largely exceeded the temporal variation in community distribution, though some structural changes were observed over time within each of them. Moreover, studies show that benthic systems do not appear to be tightly controlled by any single environmental driver, demonstrating the complexity of spatially varying temporal response (Zettler et al. 2017).

Invasive species were sometimes redrawing the major patterns of community structure in the past, and can have effects comparable with those of globalisation in human communities that, on the one hand, blur and fade the brightness of the original cultural specifics but create also new structures and functions. In the southern Baltic Sea *Marenzelleria spp.* (now significantly contributing to

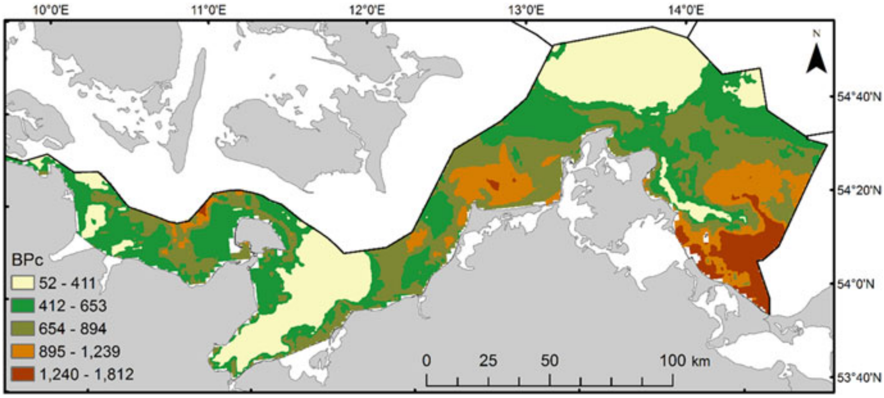

Fig. 14.2 Distribution of community bioturbation potential in the offshore waters of the German Baltic (after Gogina et al. 2020)

bioturbation in the south-western part of "Kadetrinne" and in the Pomeranian Bay) represent the invertebrate invaders.

Soft-sediment habitats largely prevail in the southern Baltic, though subtidal hard substrate and associated assemblages are also present (Beisiegel et al. 2020). For hard substrate assemblages, most of biological variability was detected across the large scale along the horizontal salinity gradient.

Better knowledge of the ecological structure of benthic macrofauna communities is urgently needed with respect to structural components, including human activities. It is essential for quantifying the flow of energy and cycling of materials through the ecosystem, informing and supporting nature-based solutions and adaptation plans and constructing accurate projections of marine ecosystem futures.

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Patterns of Bioturbation and Associated Matter Fluxes

15

Stefan Forster, Claudia Morys, and Martin Powilleit

Abstract

Bioturbation, the biogenic particle and fluid transport in sediments, is generally thought to be important in the context of matter fluxes and therefore ecosystem functions. This chapter summarizes current knowledge on bioturbation in the Southern Baltic Sea, including locally very high solute exchange. An extended study based on chlorophyll as particle tracer indicates geographic differences in functional groups of particle reworking benthic infauna. This might enhance transient retention of organic matter in shallow water sediments of slope regions. Reworking also shows high local variability and surprisingly uniform overall rates throughout the region. The text states a lack of understanding as to how indices of bioturbation (BP_C, BIP) reflect actual mechanistic functioning of animal-sediment-interactions. While displaying similar rates, bioturbation likely supports the integrity of processes occurring at and across the sediment-water interface.

The knowledge base for an assessment of particle reworking and pore water fluid exchange in sediments for functional aspects is still limited. According to a recent compilation (Solan et al. 2019), 5.5% of all studies on particle bioturbation are based on chlorophyll a as tracer. In the Baltic, many observations rely on sediment profiling imagery, particularly from the Åland Sea. Chlorophyll-based studies from the Baltic Sea are those by Morys et al. (2016, 2017) in SECOS and Powilleit

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and Forster (2018). Less than 10% of all studies determine L, the bioturbation depth, but all SECOS data do.

Worldwide, some 25% of all observations on sediment ventilation employ bromide ions as a tracer. In the Baltic Sea, studies on ventilation reflecting bioirrigation as an ecosystem function are predominantly based on bromide (Renz and Forster 2014; Powilleit and Forster 2018). Field studies on ventilation, by tracing fluid exchange with the pore waters, are scarce. An index to estimate pore water exchange based on the benthic infauna community composition (bioirrigation potential, BIP_c; Renz et al. 2018), provides a theoretical background to ventilation and bioirrigation.

Intense studies by Morys et al. (2016, 2017) analyzing up to 24 sediment cores per location allows relatively reliable interpretations. Mixing depths of 5–7 cm coincide well with worldwide results (Teal et al. 2008). Reworking coefficients, D_B, were generally in the order of <1 to 3.9 $cm^2\ d^{-1}$ and varied by a factor of 20 throughout the area. Variability was high between parallel cores within any of the locations, with apparently little difference related to location or season (please refer to Morys et al. 2016 for details of sampling locations). Regional differences were more pronounced with respect to the mode of particle reworking, an information not portrayed in the community bioturbation potential, BP_c. Between 33 and 100% of all cores from one location depicted non-local processes as an additional mode of transport, with the remaining samples described by random mixing of particles only. Non-local transport of particles occurs, for instance, in conveyor-belt feeding, when sediment is transported from deeper sediment layers to the sediment-water interface or vice versa. This non-local transport increased from Mecklenburg Bay in the West to the Pomeranian Bay in the East. Morys et al. (2017) related the non-local reworking activity to bivalves and polychaete species found within the same sediment core (*Arctica islandica, Abra alba, Limecola balthica*, *Nephtys hombergii, Scoloplos armiger*), which may thus be regarded as key species for this transport.

The close spatial association of key fauna with vertical particle transport largely fits our present process understanding, namely that these transport events should be spatially confined and their traces are transient, if observed using a reactive tracer such as chlorophyll pigments. This may be one reason for the high variation in reworking rates in the literature (Teal et al. 2008; Morys et al. 2016).

Solute exchange quantified in communities of the Pomeranian Bay (Powilleit and Forster 2018) yielded penetration depths of 10–12 cm, somewhat deeper than particle reworking activities (5–7 cm). For the Pomeranian Bight, this study reports some of the highest rates so far measured for bioirrigation, in agreement with both BIP_c hotspots (not shown) and very high BP_c (Gogina et al. 2020). This qualitative correspondence provides confidence that patterns of indices BIP_c and BP_c indeed reflect aspects of biological solute and particle transport and may serve as tools for planning and management. Still, indices are not sufficient enough to elucidate quantitative relations in biogeochemical cycling and its relation to fauna community. A great deal of understanding is still missing here (Gogina et al. 2018; Lipka et al. 2018).

In more general terms, it is widely accepted that bioturbation has relevant effects on geochemical processes, e.g. by affecting many redox-sensitive processes, facilitating co-oxidation of refractory with fresh carbon, priming and burial of carbon (Aller and Cochran 2019; Zhang et al. 2019). Bioturbation supports, for instance, C turnover (mineralization) since mixing replenishes iron and manganese oxides fueling continuous steady state microbial anaerobic processes in the sediment. In this context, Gogina et al. (2018) and Teal et al. (2010) found a relationship between the distribution of relevant pore water and particulate phase elements in sediments and BP_c or L. High mixing depths, deduced from particle bioturbation and bioirrigation, indicate the existence of a well-developed redox discontinuity layer throughout most of the investigation area.

Since particle mixing and bioirrigation are thought to be related to biomass and trait of benthic fauna or species identity, the aforementioned overall similarity in particle mixing intensity is somewhat surprising albeit with some variation. This is similar to the comparatively small changes in D_B found across large longitudinal and latitudinal regions in comparison to differences in fauna assemblage structure (Teal et al. 2008). While those authors attribute some aspects to differences in methods employed, this does not apply for the present data set (Morys et al. 2016, 2017). This observation may bear an interesting information: different communities possibly generate a rather similar level of mixing activity within a narrow bandwidth. Interestingly, biogeochemical modelling, too, assumes a rate of bioturbation on a similar order of magnitude (Radtke et al. 2019) and with little regional variation, thus reflecting the pattern shown in measured data. As a general effect, bioturbation seems to support the integrity of the sediment interface by facilitating exchange processes as they exist in a functioning coupled benthic-pelagic ecosystem.

Another general result highlights the changes in the mode of particle transport, namely its non-local component which is increasing from West to East, likely as a consequence of changing benthic community composition. Given that direct particle transport to sediment depth occurs more frequently or at higher intensity in the latter, this may have crucial consequences for the functioning of coastal sediments (Fig. 14.2). Deposition and resuspension processes at the sediment-water interface interact with random mixing by bioturbating fauna, affecting degradation of organic matter on its down-slope transport to depositional centres off the coast (Aller and Cochran 2019). Intense non-local mixing events that effectively remove some organic matter for an unknown period of time from this down-slope transport might additionally increase overall residence times of organic matter in sediments (Fig. 15.1). This "transient retention" may overall enhance organic matter decay in shallow sediments through co-oxidation and priming processes (Aller and Cochran 2019), however, it may also foster carbon preservation by exposing reactive carbon to anoxic conditions (Zhang et al. 2019; compare also case study I, Chap. 28).

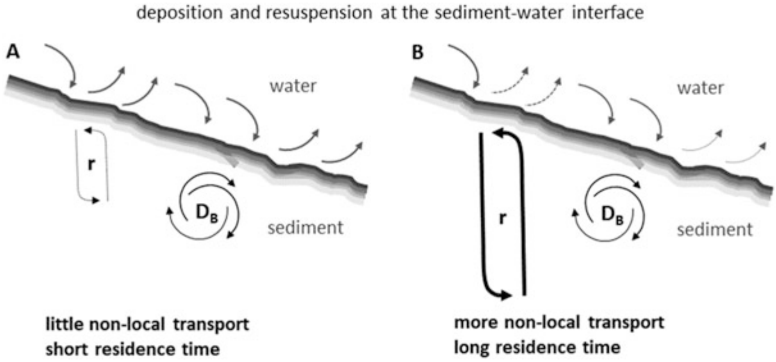

Fig. 15.1 Conceptual sketch of particulate organic matter on its way from coastal sediments to deeper depositional centres, modified by deposition-resuspension cycles and random mixing by bioturbation (D_B). Limited non-local reworking (r) may allow for relatively swift transport from shallow to deeper coastal seas (**a**), whereas intense non-local reworking may increase overall residence time of some organic matter (**b**). The latter may enhance overall organic matter decay and reduce down-slope transport (inspired by Aller and Cochran 2019)

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16 Seasonal Aspects and Short-Term Variability of the Pelagic Offshore Ecosystems

Jörg Dutz and Norbert Wasmund

Abstract

In the pelagic ecosystems, food web interactions between phytoplankton and zooplankton determine the amount of energy transferred to higher trophic levels, the remineralisation of nutrients and the export of organic material to the benthic ecosystem. The variability in the strength of this coupling has important implications for the efficiency of trophic transfer and for the match-mismatch of secondary production with higher trophic level consumers such as fish larvae. This receives particular attention in recent years with regard to potential climatic alteration of the phyto- and zooplankton phenology. In the western Baltic Sea, the recurrent seasonal patterns of phytoplankton and their short- as well as long-term variation are well understood, but little is known about the coupling to zooplankton. Data from a high-frequency sampling across the salinity gradient in the western Baltic Sea shows the well-known delay in the seasonal development of phytoplankton from the Belt Sea to the southern Baltic Proper. However, while the coupling to zooplankton in the Belt Sea is relatively tight, an increasing offset in the timing occurs in the southern Baltic Proper that affects the utilization of the spring bloom and is explained by a shift in biodiversity.

16.1 Background

The seasonal development of plankton is an annually recurrent process of community assembly and succession and is triggered by changes in external controlling factors such as temperature and light and internal biotic interactions like phytoplankton grazing by zooplankton (Kivi et al. 1993; Romagnan et al. 2015). From

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temperate to polar pelagic ecosystems, annual dynamics of phyto- and zooplankton responds with a typical pattern of their life cycle due to strong variations in environmental conditions, but substantial year-to-year variability in timing and community composition may exist. The annual course of insolation determines warming and cooling of surface waters and, subsequently, the convective mixing of the water column, as well as the viscosity of the water. It further initiates the occurrence of sea ice, precipitation and finally the input of nutrients. Together with light availability, these factors ultimately influence the development of the phytoplankton biomass and composition (Behrenfeld and Boss 2014). The spring bloom triggers the reproduction and population growth of zooplankton, which may in turn regulate the phytoplankton by its feeding activity. Resource limitation, competition, allelopathy, parasitic infections and predation are other central processes that regulate planktonic populations during the course of the year. The coupling of phyto- and zooplankton affects the energy transfer to higher trophic levels in pelagic and benthic ecosystems and influences biogeochemical cycling (Thackeray 2012; Behrenfeld and Boss 2014). Short-term anomalies in the regulatory factors can disturb the annual cycles and cause unpredictable variability. The understanding of the interannual and phenological long-term variability of pelagic plankton and its environmental control is, therefore, important for assessing the implication of climate change on the pelagic ecosystem.

The first insights into the annual succession of plankton in the western Baltic Sea date back to the pioneering study of Hensen (1887). At the beginning of the twentieth century, general patterns of species distribution and succession of phyto- and zooplankton were established from mainly seasonal studies in the Belt Sea (Kiel Bight and the Bay of Mecklenburg) and the southern Baltic Proper (Arkona and Bornholm Basins; Apstein 1906; Driver 1908; Otten 1913; Büse 1915). A high temporal resolution was achieved in Kiel Bight in the early phytoplankton studies by Lohmann (1908) and Busch (1916–1920), which were included in the more general studies of succession in the pelagic system by Smetacek (1985) and Wasmund et al. (2008). Comparably little is known about the recurrent patterns and variability of the zooplankton in the western Baltic Sea (Smetacek 1985; Zervoudaki et al. 2009). The systematic work on phytoplankton during the last 100 years revealed differences in species composition and succession between the Belt Sea and the southern Baltic Proper due to environmental settings for which the Darss Sill represents an approximate biogeographical border. In the following, the seasonal cycle in the pelagic is outlined for both areas during the year 2015. The analysis of additional material during this year, supplementing the usual low frequency monitoring, allows us to depict the seasonal succession in an unprecedented high temporal resolution, particularly for the zooplankton. While the historic biogeographic pattern of the annual cycles of phytoplankton is well represented, the analysis also reveals that biodiversity shifts caused by the salinity gradient affect the zooplankton seasonal timing and, thus, the utilization of the spring bloom production.

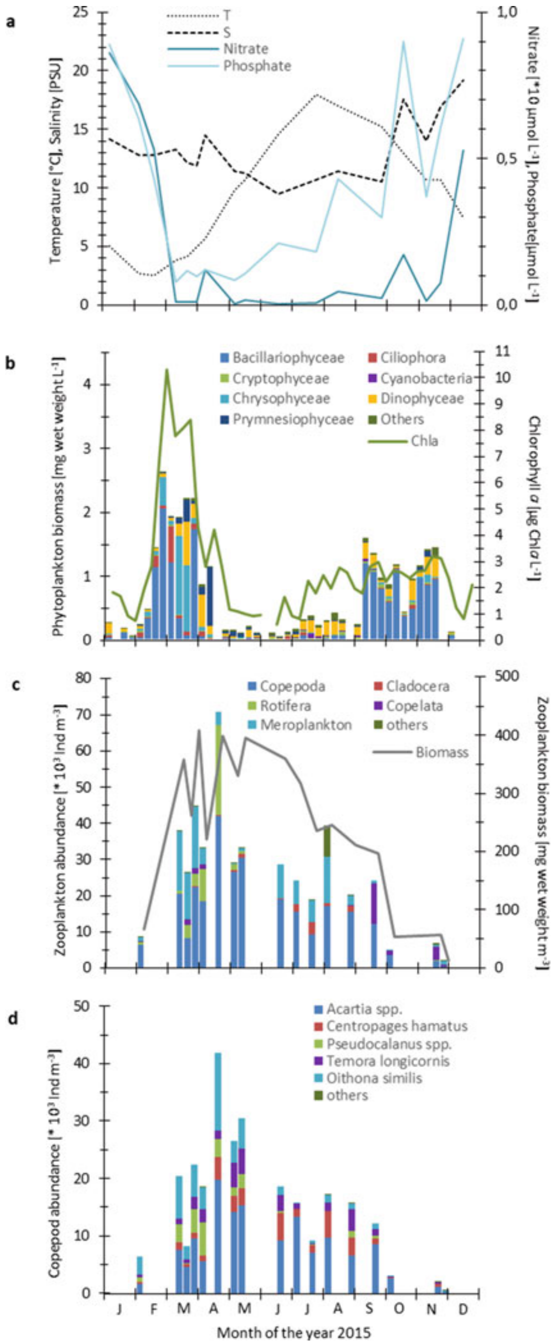
16.2 Belt Sea

The seasonal succession in the pelagic zone of the shallower Belt Sea is illustrated by the temporal development of the physical conditions, nutrient concentrations and the plankton biomass, abundance and composition in the Mecklenburg Bight in 2015 (Fig. 16.1). The hydrological conditions in the Belt Sea can vary considerably due to the variable in- and outflow of saline and brackish water, respectively, or wind-induced mixing (Lenz 1977) and can have a strong effect on the nutrient conditions (von Bodungen 1986). Nevertheless, mixing, advection and the pronounced seasonal warming cause a regular variation in the environmental conditions with a maximum in salinity during winter and a summer maximum of temperature ($S = 9.5$–19.2 PSU, $T = 2.5$–17.9 °C, Fig. 16.1a).

Winter stocks of phyto- and zooplankton are generally low and nutrient levels are high (Fig. 16.1a–c). Calanoid and cyclopoid copepods dominate the small stock of zooplankton and a high proportion of nauplii indicate active overwintering. The phytoplankton spring bloom usually develops with increasing light intensity and when light penetrates the upper mixed layer, i.e. reaches the bottom or the permanent halocline in the shallow water. The timing is mainly controlled by weather conditions (Smetacek 1985; Wasmund and Siegel 2008). In calm and sunny periods, the spring bloom may already start in February, whereas in windy and cloudy springs, the bloom is delayed to March. In the Belt Sea, it is always dominated by diatoms such as *Skeletonema marinoi*, *Chaetoceros* spp. and *Rhizosolenia* spp., but their contributions vary strongly from year to year. Although the water temperature is still low, zooplankton begins to increase due to hatching of resting eggs in sediments and the initiation of reproduction of copepods by the spring bloom (Fig. 16.1c). Typically, meroplankton contributes considerably to the zooplankton due to early spawning of benthic polychaetes during early spring. Because of low consumption rates of the small development stages of copepods at low temperature and the partly lecithotrophic larval development in meroplankton the grazing rates of the zooplankton community are presumably low and unlikely to control the spring bloom development (Fig. 16.1).

The diatom bloom terminates by the nutrient depletion in the upper mixed layer (Fig. 16.1a) and rapid sedimentation of the diatom cells. Flagellates usually follow the diatom bloom as they may use low ambient concentrations of quickly recycled nutrients or are able to perform vertical migrations in order to acquire nutrients which are still available in deeper water layers. These flagellate blooms may be more or less clearly separated from the diatom bloom (Fig. 16.1b) and are referred to as 'late spring bloom' or 'post-spring bloom' (Smetacek 1985; Wasmund et al. 2008). Dinophyceae such as *Peridiniella danica* and Gymnodinales dominate in March–April together with the dictyochophyte *Dictyocha speculum*. In the late phase, exceptional blooms of Prymnesiophyceae may follow, such as *Chrysochromulina* in 2015. The zooplankton abundance and biomass increase to the annual maximum during this period. The increase is typically associated with continuous reproduction of copepods and maturation of developing cohorts of *Pseudocalanus* spp., *Oithona similis* and *Acartia longiremis*, which tolerate the still low water temperatures

Fig. 16.1 Seasonal variation of the hydro-chemical variables in the surface water layer and the succession of phyto- and zooplankton in the Bay of Mecklenburg (Belt Sea) in 2015. (**a**) Temperature (T), salinity (S), nitrate and phosphate, (**b**) taxonomic composition of the phytoplankton and total biomass (Chl$a$), (**c**) zooplankton biomass and composition of major taxonomic groups, (**d**) taxonomic composition of the copepods



(Fig. 16.1c and d). Polychaete larvae have vanished, but occasionally rotifers may occur. The zooplankton exerts a strong control of the phytoplankton due to its high biomass and possesses a vital function in the recycling of nutrients (Smetacek 1985).

Phytoplankton biomass is usually low after the post-spring bloom (Fig. 16.1b). Depending on sporadic storm-induced injections of nutrient-rich bottom water to the upper layer, however, a highly diverse summer community develops. It is usually dominated by large-celled diatoms (e.g. *Dactyliosolen fragilissimus*, *Guinardia flaccida*, *Proboscia alata*, *Cerataulina pelagica*) and dinoflagellates (*Ceratium* spp.). The magnitude and composition of this summer bloom are highly variable and can be missing as in 2015. During this period, the zooplankton composition shifts due to the warming of the water column (Fig. 16.1a and c). Copepods become less frequent and *Pseudocalanus*/*Oithona* declines in favour of *Temora longicornis*, *Centropages hamatus* and *Acartia bifilosa*. Meroplankton can be abundant again, this time dominated by bivalve larvae. Occasionally, large numbers of Tintinnida can be observed and carnivorous marine cladocera of the genera *Evadne* and *Podon* occur in low numbers.

The large-celled diatoms and dinoflagellates are hardly grazed and may further grow during summer stratification based on efficient nutrient recycling. Supported by nutrients from deeper water layers after the breakdown of the summer stratification, large autumn blooms may develop. In the Belt Sea, the *Ceratium* autumn bloom is the most recurrent feature of phytoplankton succession that has been observed since the last century (Lohmann 1908; Busch 1916–1920; Smetacek 1985). Nutrient input may further promote blooms of diatoms such as *Rhizosolenia*, *Cerataulina* or *Proboscia* besides *Ceratium*. The summer zooplankton biomass typically declines during this period in response to autumn cooling (Fig. 16.1c). The spring-summer copepod species *A. bifilosa* and *A. longiremis* are typically replaced by *Acartia tonsa*. The microphagous appendicularian *Oikopleura dioica* is also a prominent member of the zooplankton community in late autumn of the Belt Sea.

16.3 Southern Baltic Proper

The mechanisms and pattern of seasonal succession are principally the same in the Belt Sea and the southern Baltic Proper. However, differences in the hydrological characteristics have strong implications for the biodiversity and the seasonal development of the pelagic ecosystem of the open waters of the Arkona and the Bornholm Basin. Due to the larger water depth and distance from the shore, the upper mixed water layers are less influenced by river run-off, upwelling or major Baltic inflows of saline water. The salinity of the brackish surface water is, therefore, lower and varies less than in the Belt Sea, while the strong seasonality in seawater temperature persists ($S = 7.9$–9.8 PSU, $T = 3.8$–17.6 °C, Fig. 16.2a).

In contrast to the shallow Belt Sea, in the Baltic Proper the spring bloom may develop only after a thermal stabilization of the upper water layers. A weak stratification by warming of the upper water layers occurs already long before the establishment of the seasonal thermocline. However, the time until the upper mixed layer

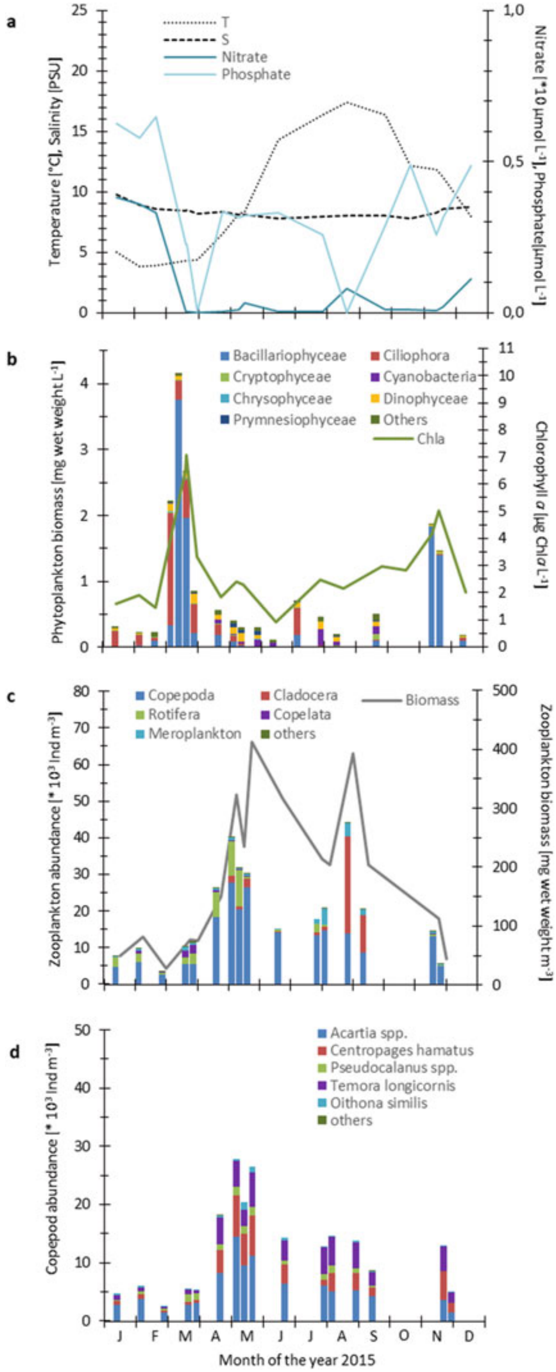


Fig. 16.2 Seasonal variation of the hydro-chemical variables in the surface water layer and the succession of phyto- and zooplankton in the Arkona Basin (southern Baltic Proper) in 2015. (**a**) Temperature (*T*), salinity (*S*), nitrate and phosphate, (**b**) taxonomic composition of the phytoplankton and total biomass (Chl*a*), (**c**) zooplankton biomass and composition of major taxonomic groups, (**d**) taxonomic composition of the copepods

is exceeded by the euphotic zone, and until the phytoplankton regularly receives sufficient light for growth, is delayed in the deep basins of the Baltic Proper. The start of the spring bloom is, therefore, retarded into eastern direction and does not occur earlier than March (Wasmund et al. 1998; Groetsch et al. 2016, Fig. 16.2b). The spring bloom was originally composed of diatoms (*Thalassiosira* spp., *Skeletonema marinoi*), but increasing contributions of dinoflagellates, e.g. *Peridiniella catenata*, occur since the late 1980s (Wasmund 2017) whereas the mixotrophic ciliate *Mesodinium rubrum* increased strongly in the late 1990s. In 2015, an early spring bloom of *Mesodinium* was followed by a *S. marinoi* peak in mid-March in the Arkona Basin (Fig. 16.2b), while *M. rubrum* dominated the entire spring bloom in the Bornholm Basin.

The timing of the zooplankton spring development is remarkably delayed in the southern Baltic Proper. The abundance and biomass remain low during the spring bloom in March and increase only during the post-bloom phase in April–May, which is a month later compared to the Belt Sea (Fig. 16.2c). A later recruitment and growth of zooplankton in response to the later onset of the phytoplankton spring bloom naturally contribute to the postponed increase. However, the delay reflects also a shift in biodiversity from the Belt Sea to the southern Baltic Proper. The cold water adapted stenohaline copepod species such as *Pseudocalanus* spp. and *Oithona similis* have largely vanished in response to the reduced salinity in the southern Baltic Proper. Since they build up, together with the polychaete larvae, the early peak in zooplankton biomass in the Belt Sea, the seasonal increase in the southern Baltic Sea is based on more tolerant copepod species such as *T. longicornis*, *C. hamatus* and *A. bifilosa/longiremis* (Dutz et al. 2010; Dutz and Christensen 2018) and rotifers that are occurring later in the season (Fig. 16.2c).

The spring bloom is terminated by nutrient limitation (Fig. 16.2a). Inorganic nutrients, primarily DIN, are exhausted in the water column down to the halocline and nutrients below this permanent pycnocline are not accessible even by migrating phytoplankton. The copepods *T. longicornis*, *C. hamatus* and *A. bifilosa/longiremis* dominate zooplankton biomass. Cladocerans like *Evadne nordmanni* and *Podon leuckartii* and the rotifer *Synchaeta* spp. occur usually in low numbers, but interannual variability exists.

As phosphate is still available in low concentrations after the spring bloom, typically nitrogen-fixing cyanobacteria develop. They are independent of nitrogen compounds as they may use dinitrogen (N_2) directly. Blooms can strongly vary from year to year especially in the Arkona Basin. They usually start with *Aphanizomenon* sp. and continue by *Nodularia spumigena* or *Dolichospermum* spp. in July and August until their termination by phosphorus limitation or strong winds. They deliver nitrogen for a moderate phytoplankton growth in summer. During this time, a second, late summer peak in zooplankton biomass usually develops with the occurrence of the brackish water cladoceran *Bosmina* spp. (Fig. 16.2c). The genus is capable of parthenogenetic reproduction and can achieve very high concentrations up to a million individuals per m^3 in relative short time periods. At this time, larvae of bivalves can also be abundant together with the copepods

Temora, *Centropages* and *Acartia*. *A. longiremis* and *A. bifilosa* are usually not replaced by *A. tonsa* likely due to the larger distance from coastal areas.

The outburst of the autumn bloom is initiated by the breakdown of the summer stratification in late September, resulting in upward mixing of nutrients from stagnant bottom water. Diatoms benefit from these nutrient pulses and blooms of large-celled species such as *Coscinodiscus granii* occur. The zooplankton biomass is still considerably large due to the prevalence of *T. longicornis* and *C. hamatus* in the area. The species actively overwinter in the water column and benefit from the autumn phytoplankton bloom. *Acartia* spp., in contrast, vanishes since these species overwinter as resting eggs in the sediment (Katajisto et al. 1998) or in a state of reduced activity (Norrbin 1996).

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Long-Term Trends of the Offshore Ecosystems

17

Norbert Wasmund and Michael L. Zettler

Abstract

Long-term data sets are crucial in assessing the state of the marine systems and its ecological processes, to disentangle human-induced and natural changes, short-term fluctuations and long-term trends. A clear trend was observed in phytoplankton composition. The dominant phytoplankton classes in the Baltic Sea, diatoms and dinoflagellates, showed an opposing trend in the spring bloom of the open Baltic Proper. Diatoms decreased and dinoflagellates increased suddenly since the late 1980s. Nearly at the same time, also a shift in the macrozoobenthos occurred in the southern Baltic Sea. The biocenotic shift in the second half of the 1990s for various members of the food chain, both in pelagic and benthic habitats, is apparently a widespread phenomenon, as it has been observed even in the eastern North and Central Atlantic. It represents probably a second ecosystem regime shift within the investigation period, which is less remarkable and less known than the first one, but nevertheless needs attention.

The Baltic Sea is heavily impacted by natural and anthropogenic pressures such as eutrophication (HELCOM 2015), climate change (BACC 2015), pollution by hazardous substances (HELCOM 2018b) and invading species (Olenina et al. 2010). In contrast to far more eutrophicated coastal lagoons, which show a "paradox of enrichment" (see Chap. 13), the excessive nutrient input from the densely populated and intensely cultivated catchment area has induced an increase in primary production, phytoplankton biomass and turbidity in the euphotic zone of the offshore ecosystems, with oxygen deficit in deep water layers since the 1950s (Andersen et al. 2017; Murray et al. 2019). The changing environmental conditions provoke

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changes in the biotic components of the ecosystem and their interactions. In order to trace the developments in this ecosystem, primarily the eutrophication, a monitoring programme was launched by the Baltic Marine Environment Protection Commission (Helsinki Commission, HELCOM) in the Baltic Sea in 1979. The Marine Strategy Framework Directive (MSFD) of the European Union (European Commission 2008) demands to combat detrimental trends and to reach a "good environmental status" (GES) in European marine waters. The analyses of trends in the different environmental parameters may indicate whether the aims of the MSFD are reached or not.

Since the HELCOM monitoring is performed according to mandatory manuals by all riparian countries (HELCOM 2017), the data from all contracting parties are consistent and are stored in the ICES database. They may form a comprehensive basis for trend analyses of phyto- and zooplankton conducted in different regions of the Baltic Sea, e.g. in its northern parts by Jurgensone et al. (2011) and Suikkanen et al. (2013). Henriksen (2009) analyzed trends in the Kattegat/Belt Sea area and the Arkona Basin. The analyses by Möllmann et al. (2009), Olli et al. (2011) and Wasmund et al. (2011) cover the whole Baltic Proper. In this book, we concentrate on trends and regime shifts in the southern Baltic Sea.

17.1 Phytoplankton

Phytoplankton biomass is represented by the chlorophyll *a* (chl-a) concentrations in the suspended particulate matter in the water, which is a core indicator in the MSFD, reflecting eutrophication. The MSFD uses summer chl-a concentrations as these data are less variable than spring data. However, we think that spring data are more directly linked to nutrient inputs, which occur mostly in winter and spring due to water erosion of the bare soil (new nutrients) and therefore more related to eutrophication than summer nutrients, which are recycled (regenerated nutrients). We abstain from showing the summer data because they are already presented in Appendix 2 of HELCOM (2018a). Spring data reveal strong trends in contrast to summer data, as demonstrated by Wasmund and Siegel (2008). We extended the data used by Wasmund and Siegel (2008) and coastal stations are added to the central Baltic Monitoring programme stations (BMP-stations K2, K4, K5, K7, K8, M1, M2) if they are not situated in inner coastal lagoons ("Boddens") and are not influenced by them. These are stations O5, O22 and HD ("Heiligendamm") in the Bay of Mecklenburg, O9 and O11 in the Arkona Basin and K12 in the Bornholm Basin (Fig. 17.1). The 3-year moving averages, including the year before and after the year indicated, are shown in Fig. 17.2. Data of Arkona and Bornholm Basins were combined as these basins reveal similar trends (cf. Wasmund and Siegel 2008; Wasmund et al. 2011).

The nutrient (N, P) input has decreased since the 1980s (HELCOM 2015), but chl-a concentrations still increased in the Arkona and Bornholm Basins because of a high internal nutrient store in the system, which was demonstrated by Conley et al. (2002). However, also intermittent periods of decrease occurred (Fig. 17.2b). The high peak values of chl-a in spring of 2010 and 2011 in the Arkona Basin coincided

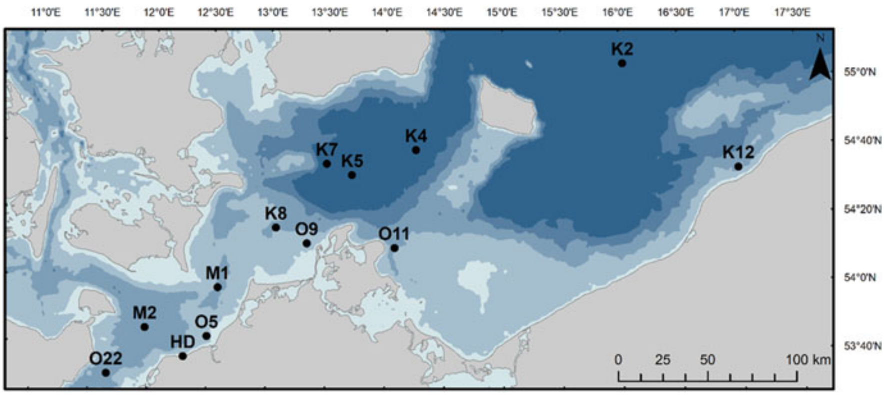

Fig. 17.1 Location of stations in the Southwestern Baltic referred to in this chapter. These include stations of the Baltic Monitoring Programme (BMP), national monitoring at locations closer to the coast, and one location at Heiligendamm (HD) monitored by the IOW

with cold preceding winters (cf. Wasmund et al. 2013 for winter minimum temperatures; see also Hjerne et al. 2019 for Swedish waters). Also in the open Bay of Mecklenburg, these years were characterized by extremely high spring bloom peaks (about 18 µg L^{-1} chl-a). Astonishingly, the spring data from the Bay of Mecklenburg revealed a strong negative trend at the beginning of the time series (Fig. 17.2a). Decreasing chl-a trends were reported in Kiel Bay from 1988 to 2012 by Lennartz et al. (2014). The contrasting trends, i.e. decrease in the Bay of Mecklenburg and increase in the Baltic Proper, were also detected in the microscopically determined phytoplankton biomass data (Wasmund et al. 2011). Although data of the 1980s might have represented the spring blooms well, they excluded most of the pre- and post-bloom period, whereas more of these data were included in the last 30 years. In this period data frequency strongly improved, especially by the weekly coastal monitoring at station "Heiligendamm" conducted since 1989. Thus, since the average number of data in Bay of Mecklenburg increased from 3.5 in the 1980s to 20.2 in the period after 1989 (compare Fig. 17.2a), the data series may be insufficient for a reliable trend analysis.

According to Olli et al. (2011), changes in phytoplankton are not clearly associated with eutrophication. Suikkanen et al. (2013) found that temperature had a greater effect than eutrophication. Already Edwards et al. (2006) wrote that bloom events may be incorrectly attributed to eutrophication, while the real modifier of change could be climatic in origin.

A clear trend was observed in phytoplankton composition. The dominant phytoplankton classes in the Baltic Sea, diatoms and dinoflagellates, showed an opposing trend in the spring bloom of the open Baltic Proper. Diatoms decreased and dinoflagellates increased suddenly since the late 1980s (Wasmund et al. 1998, 2013; Hjerne et al. 2019). The concurrent replacement of diatoms by dinoflagellates was also found in other regions of the northern hemisphere, like the North Sea (Reid et al. 2001; Weijerman et al. 2005), Mediterranean Sea (Goffart et al. 2002), North

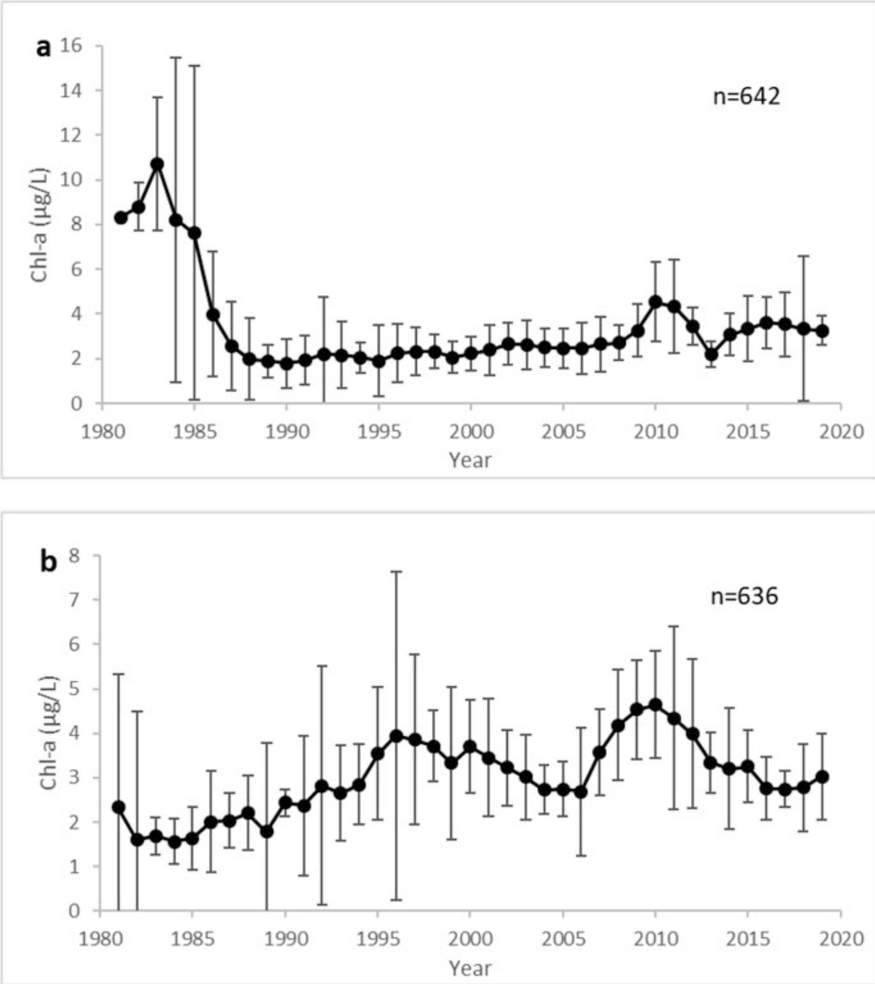

Fig. 17.2 Trends in spring chl-a concentrations in the upper water layer of (**a**) the Bay of Mecklenburg (February–April) and (**b**) the combined data of Arkona and Bornholm Basins (March–May), based on open sea monitoring stations and outer coastal stations not influenced by lagoon and river water. For smoothing, the 3-year moving averages are shown. Confidence intervals for alpha = 0.05 are given. Please note the different scales

Atlantic Ocean (Choi et al. 2005) and North Pacific Ocean (Hare and Mantua 2000). Simultaneous shifts have already been reported for phytoplankton, zooplankton, fish (Alheit et al. 2005; Möllmann et al. 2009) as well as for zoobenthos (Kröncke et al. 2013; Zettler et al. 2017) in the Baltic Sea. They may be called an "ecological regime shift", which is typically characterized by infrequent and abrupt changes in ecosystem structure and function, occurring at multiple trophic levels and on large geographic scales (Möllmann et al. 2009).

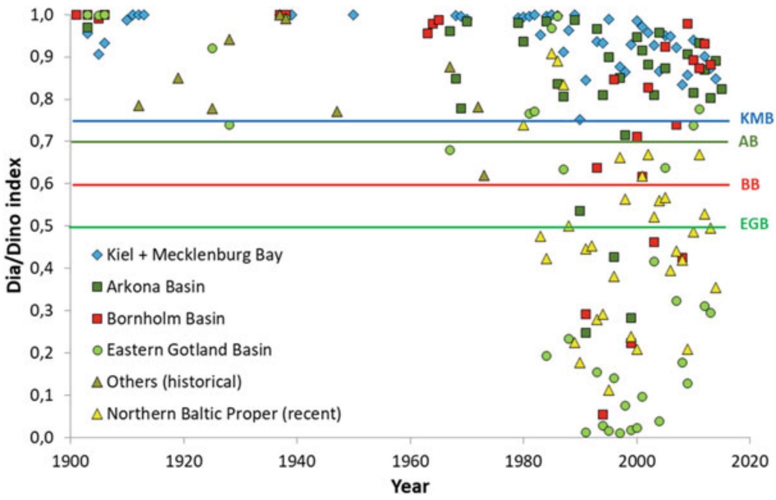

Fig. 17.3 Summary of historical and recent Dia/Dino indices in different areas of the Baltic Sea since the year 1901. Suggestions of limits for GES for Kiel Bay and Bay of Mecklenburg (KMB), Arkona Basin (AB), Bornholm Basin (BB) and Eastern Gotland Basin (EGB) are plotted as lines (Reproduced from Wasmund 2017)

The shift from diatoms to dinoflagellates in the spring blooms of the Baltic used to define an indicator suggested for the implementation of the MSFD. This diatom/dinoflagellate index (Dia/Dino index), considers the biomass ratio of diatoms and dinoflagellates during the spring period (Wasmund et al. 2017). This index was strongly decreasing from a value of almost 1.0 (which means complete diatom dominance in relation to dinoflagellates) to nearly zero (which means complete dinoflagellate dominance in relation to diatoms) in the Eastern Gotland Basin and the Bornholm Basin, while in Mecklenburg Bay and Kiel Bay (Fig. 17.3) it fell below the suggested threshold for GES in the Baltic Proper. The decrease in diatoms was correlated with milder winters since 1988/89 (Wasmund et al. 1998; Kotta et al. 2018; Hjerne et al. 2019). A conceptual model for explanation of the temperature control of the diatom or dinoflagellate spring blooms is suggested by Spilling et al. (2018). Wasmund et al. (2013) discussed a lack of deep mixing after mild winters (stratification hypothesis) and a stronger top-down regulation of the early diatom bloom by earlier developing zooplankton after mild winters (food web hypothesis). Also the missing diatom decline in the Belt Sea may be explained by these hypotheses (Wasmund et al. 2013).

The slight recovery of the Dia/Dino index approximately after 1998 despite still mild winters may be explained by a strong increase of the mixotrophic ciliate *Mesodinium rubrum* which increasingly became dominant especially in spring at the end of the 1990s (Wasmund et al. 2011). This motile species seems to cover a similar ecological niche as the dinoflagellates of spring (e.g. *Peridiniella catenata*) and suppresses them, which is reflected in an increase of the Dia/Dino index. Also Klais et al. (2011) reported a decreasing proportion of dinoflagellates versus diatoms in the southern Baltic Sea from 1995 to 2004. Even in the North Sea, a decrease in dinoflagellates together with a decrease in copepod abundance was identified after 1998 by Alvarez-Fernandez et al. (2012). Nearly at the same time, also a shift in the macrozoobenthos occurred in the southern Baltic Sea, as discussed below. The biocoenotic shift in the second half of the 1990s is obviously a widespread phenomenon as it was observed even in the eastern North and Central Atlantic (Alheit et al. 2014). Moreover, a "climate regime shift" in 1998 is described for the western North Pacific (Zhao et al. 2018). It represents probably a second ecosystem regime shift within the investigation period, which is less remarkable and less known than the first one, but nevertheless needs attention.

A phytoplankton group of socio-economic relevance is that of cyanobacteria (compare also Chap. 28, Case Study III). The nitrogen-fixing cyanobacteria form blooms of unpleasant appearance and therefore may impair tourism that has high economic importance in the coastal regions. Moreover, they are toxic and may cause incidents (Wasmund 2002). By their ability for nitrogen fixation, they counteract measures to combat eutrophication (Vahtera et al. 2007). Long-term analyses including historical data revealed that cyanobacterial blooms became a common phenomenon since the 1960s (Finni et al. 2001). However, cyanobacteria data are highly variable because of the high patchiness of the blooms. Öberg (2017) showed high fluctuations from year to year for cyanobacteria data based on satellite images from 1997 to 2016. Kahru and Elmgren (2014) discovered in a satellite time series a significantly higher areal fraction with cyanobacteria accumulations for the second half of the time series (1997–2013) than for the first half (1979–1996). The frequency of accumulations of cyanobacteria at the sea surface was correlated with phosphorus concentrations and water temperature in the surface layer (Kahru et al. 2020). The monitoring data starting in 1979 revealed, however, a decrease in bloom-forming cyanobacteria, particularly in *Aphanizomenon* sp., in the southern Baltic Proper, in contrast to northern regions of the Baltic Sea (Wasmund et al. 2011; Olofsson et al. 2019). Nevertheless, cyanobacteria seem to increase on a worldwide scale due to global warming (Karlberg and Wulff 2013; Paerl and Otten 2013).

Another trend concerns the phenology, which is the phenomenon of earlier start of the growing season. Based on weekly samples from the coastal station "Heiligendamm" from 1988 to 2017, Wasmund et al. (2019) discovered an earlier start of the phytoplankton spring bloom at a rate of 1.4 d a^{-1} and even a delay of the end of the autumn bloom by 3.1 d a^{-1}. The growing season was assumed to start if a biomass or chl-a threshold was reached for the first time in a year and it probably ends when chl-a sinks below the threshold value a last time. Its duration increased from 159 days in 1989 to 284 days in 2017 if based on microscopically determined biomass data (Utermöhl method, cf. HELCOM 2017) and from 163 days in 1989 to 292 days in 2017 if based on chl-a data (Fig. 17.4). The earlier start of the growing season was correlated with a slight increase in sunshine duration during spring whereas the later end of the growing season was correlated with a strong increase in water temperature in autumn. The extension of the growing season did not necessarily lead to higher annual phytoplankton production or biomass because

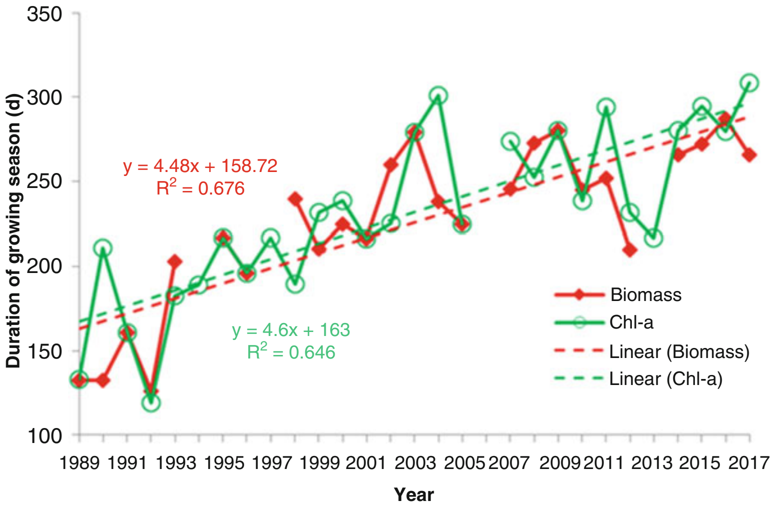

Fig. 17.4 Trends in the duration of the phytoplankton growing season at station "Heiligendamm", based on biomass data and chl-a data, with regression lines and corresponding formulas (Reproduced from Wasmund et al. 2019)

the spring and autumn blooms did not extend but only shift to earlier and later dates, respectively, which lead to a prolongation of the summer biomass minimum.

Using satellite-estimated chl-a data, Kahru et al. (2016) estimated even an extension of the phytoplankton growing season from 110 d in 1998 to 220 d in 2013 in the central Baltic Sea. A general trend of earlier phytoplankton spring blooms by 1–2 weeks over the last 20 years, associated with more sunshine and less windy conditions, was also described by Hjerne et al. (2019) in Swedish waters of the northern Baltic Proper.

17.2 Macrozoobenthos

The shift from diatoms to dinoflagellates and *M. rubrum* is expected to increase the energy transfer to pelagic secondary production and decrease spring bloom inputs to the benthic system (Spilling et al. 2018; Hjerne et al. 2019). Therefore, the Dia/Dino index is primarily a food web indicator, influencing the zooplankton and the macrozoobenthos by changes in food quantity and quality. Correspondingly, the regime shift of the late 1980s could be observed also for benthic communities in several regions of the Baltic Sea (e.g. Laine et al. 1997; Rousi et al. 2013; Zettler et al. 2017; see below). Other long-term analyses showed no clear shifts or trends but rather variability of the composition and abundance (e.g. Dippner and Ikauniece 2001) or were connected with recurring hypoxic events and with the large-scale expansion of non-indigenous species (e.g. Maximov 2015).

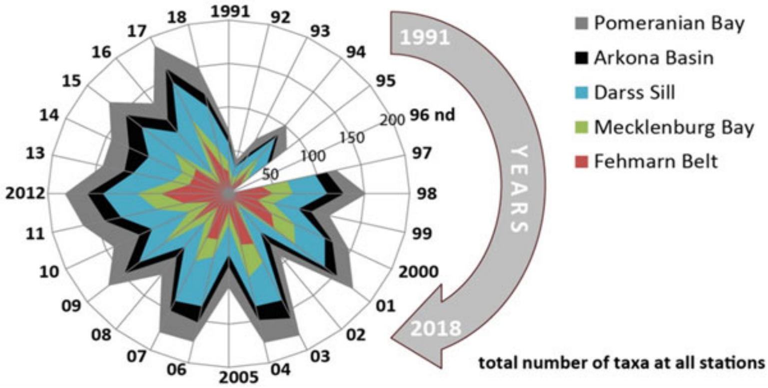

Fig. 17.5 Long-term variability in macrobenthic species richness at five stations along the German Baltic Sea coast from 1991 to 2018 (*nd* no data)

Eutrophication has been an increasing ecological threat during the past 50 years in many Baltic marine waters (Karlson et al. 2002). In all depth ranges a threefold–tenfold increase of macrozoobenthos abundance from the past to the present has been observed that might have been caused by eutrophication (Zettler et al. 2006). For the Pomeranian Bay, Kube et al. (1997) could show similar results. Eutrophication caused an increased biomass of filter-feeding bivalves in the shallower water depths, but oxygen depletion decreased the species richness at stations deeper than 15 m. However, no obvious trend could be detected. In fact the changes reflect high variability in time and space rather than a shift. For example the distribution patterns of several species of the Arkona Basin have changed several times during the last 80 years (Zettler et al. 2006). Nevertheless, with few exceptions no consistent change from the past to present days could be observed. Species composition of the macrozoobenthic community in the shallower area was very similar during all time periods. Only in deeper waters we found differences, which may be explained by past changes in hydrography (e.g. salinity, oxygen).

According to the long-term data from a large geographical range of the southern Baltic Sea, strong and significant variability could be observed during the last three decades (Fig. 17.5). Depending on the salinity gradient and the oxygen regime, the diversity was different between years and regions. With 40 species at all stations, the lowest “year-value” was observed in 1992. In contrast the highest diversity occurred in the year 2017 with about 189 species. Regarding the region the diversity was usually highest in the high-saline westernmost area at Fehmarn; however, strong oxygen depression had reduced the species diversity in the years 2002, 2005, 2008 and 2016 significantly.

Clear changes within the macrozoobenthos component could be observed in some areas of the southern Baltic Sea (Zettler et al. 2017). Multivariate analysis was employed to visualize trends in community composition. While for the muddy stations in the Mecklenburg Bight and Arkona Basin with frequent influence by oxygen deficiency no clear trends could be observed, the benthic community at the

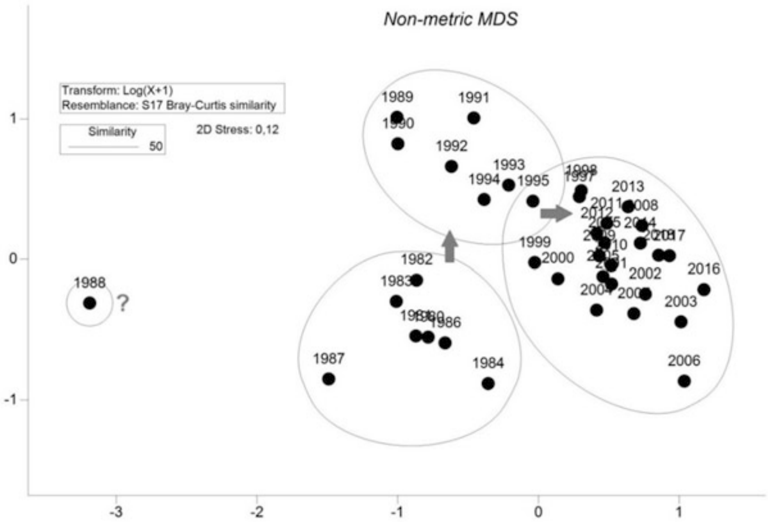

Fig. 17.6 Ordination of Bray-Curtis similarities in species composition and abundance of the station at the Darß Sill from 1980 to 2018. Arrows indicate strong changes and might be seen as "regime shifts" (Adapted from Zettler et al. 2017)

Darß Sill shifted from one quasi-stable state to a next state at the end of the 1980s (Fig. 17.6). This agrees with the North Atlantic regime shift found in phytoplankton, zooplankton and fish (see above, Fig. 17.3). A second shift occurred in the mid-1990s.

Long-term data sets are crucial in assessing the state of the marine systems and its ecological processes, to disentangle human-induced and natural changes, short-term fluctuations and long-term trends (Rousi et al. 2013; Dippner et al. 2014; Haase et al. 2016; Zettler et al. 2017). Natural and anthropogenic factors influence the variability of the biological environment simultaneously and it is always a challenge to discover the real drivers and pressures and to derive an appropriate management strategy.

Acknowledgement Besides the original phytoplankton data from the Leibniz Institute of Baltic Sea Research, also foreign phytoplankton data from the ICES Dataset on Ocean Hydrography (The International Council for the Exploration of the Sea, Copenhagen) were used.

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18 Nutrient and Limitation Regimes in Coastal Water Ecosystems

Maximilian Berthold

Abstract

Phytoplankton blooms are an ongoing issue in coastal waters, especially if the catchment area is under strong anthropogenic pressure and nutrient flows are high. Restoring such coastal waters can be difficult, as baselines for nutrient demand of the dominant primary producers may have shifted with eutrophication. Monitoring data can be used to describe the overall nutrient regime in coastal waters and occurring seasonal limitation patterns. Here, I identified six distinct phytoplankton bloom patterns at 31 monitoring stations of the German Baltic coastal waters using changepoint-analyses and wavelet-transformations. There were no detectable trends over 20 years, but an impact of extreme weather events on phytoplankton biomass was observed. Furthermore, a generalized additive mixed model allowed for the identification of seasonal changing nutrient availability, in part explained by the onset or presence of phytoplankton blooms. In conclusion the phytoplankton bloom type of a coastal water could drive seasonal patterns of nutrients, but was not necessarily described by it, as other factors like grazing impact are currently not monitored.

18.1 Background

Most German coastal water ecosystems showed increased nutrient supply, resulting in frequent phytoplankton blooms and deteriorated submerged macrophyte stands (Schiewer 1998; Blümel et al. 2002; Berthold et al. 2018a). Consequently, most German coastal waters did not reach a "good ecological state" in 2016, as requested

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by the EU-Water framework directive (European Community 2000; LUNG 2013). The “good ecological state” is defined by discrete nutrient thresholds derived from maximum allowable nutrient inputs into German coastal waters (HELCOM 2013). Eutrophication driving factors seem to differ regionally as well, as indicated by lower chlorophyll: nutrient ratios in Danish compared to German coastal waters with similar salinity conditions (Kronvang et al. 2005). Nonetheless, stakeholders need evidence-based knowledge to counteract eutrophication processes. The description of periodicities in inter- and intra-annual nutrient development and, consequently, limitation regimes for phytoplankton could be a possible approach to fill the current knowledge gap.

18.2 Data Basis

German coastal waters have been closely monitored during the past decades. The German state agencies of Mecklenburg-Vorpommern (State agency for Environment, Nature Conservation and Geology–LUNG) and Schleswig-Holstein (State agency for Agriculture, Environment and Rural Areas—LLUR) monitor Baltic coastal waters monthly for several abiotic and biotic factors. The chapter focuses on the coastal development 10 years after improved water treatment plants were installed, to analyze current trends and effects in human-managed systems. I, therefore, used a subset of the LUNG dataset ranging from 2000 to 2018, including 31 stations in southern German coastal waters. Five stations were within 2 km of the outer coastline (all mesohaline, type B3), whereas the remaining stations were estuarine and marine lagoons or shallow bays ranging from oligo- (type B1) to mesohaline (type B2) water. I used Chlorophyll *a* (Chl *a*, n = 5561), dissolved inorganic nitrogen (DIN, sum of NO_3, NO_2, NH_4, n = 5417), dissolved inorganic phosphorus (DIP, $n = 5489$), silica (SiO_4, $n = 5342$), total nitrogen (TN, $n = 5495$), and total phosphorus (TP, $n = 5514$). All variables were from the first two meters of the water column. Statistics were done in R 3.5.0 (R Core Team 2019).

18.3 Coastal Bloom Types and Spatial Trends

As a first step, a changepoint-analysis (R-package *changepoint*, Killick et al. 2014) was applied with monthly Chl *a*-medians of each station (method binary segmentation “BinSeg”, number of possible changepoints = 7). This approach allowed to define a new grouping variable “bloom type”, based on phytoplankton bloom timing within a year. I followed the suggestion of Carstensen et al. (2015) that there is no universal bloom definition, but that a bloom is a substantial deviation from background phytoplankton biomass. The changepoint-output was used to calculate a hierarchical dendrogram based on median Chl *a*, TN, TP, DIN, DIP and SiO_4 concentrations. The changepoint-analysis and dendrogram analyses revealed that all 31 observed stations can be grouped in at least six clusters (Fig. 18.1). Bloom

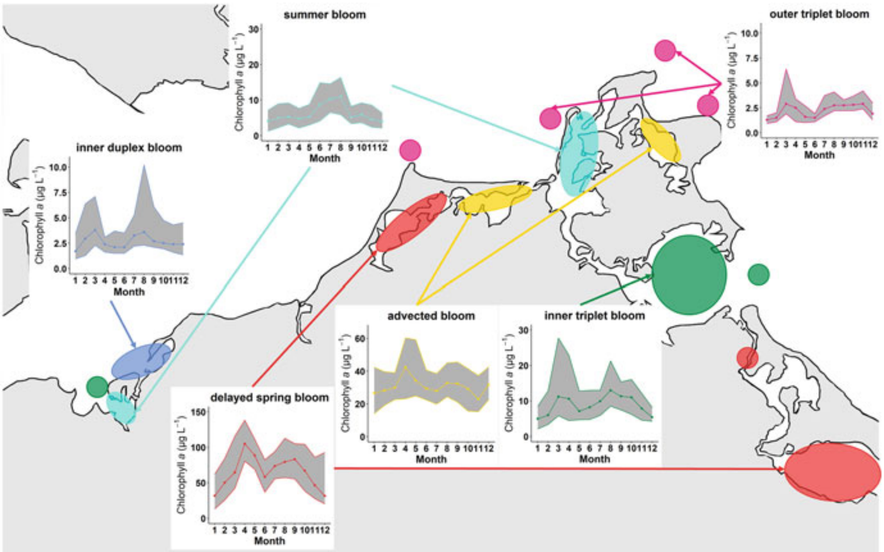

Fig. 18.1 Chlorophyll-concentrations per bloom type phenotype along the southern Baltic Sea coast. Arrows indicate positions of the respective bloom type. Dots in the plots represent long-term monthly median values, ribbons represent 25–75% monthly quantiles. Outer triplet bloom, $n = 658$, four stations; inner duplex bloom, $n = 582$, three stations; summer bloom, $n = 1127$, six stations; inner triplet bloom, $n = 1241$, six stations; advected bloom, $n = 906$, five stations; delayed spring bloom, $n = 1185$, six stations. Plots were created with R-package *ggplot2* (Wickham 2016)

type patterns did not cluster within the same geographic region, pointing to driving factors independent of sampling location (Fig. 18.1).

All stations of the outer coastal waters except one (no. O133) showed three distinct Chl *a* peaks in March, August and October ("outer triplet bloom"). Three sampling stations in the bay of Wismar with higher salinity clustered together showing two blooms with higher Chl *a* concentrations in autumn than in spring ("inner duplex bloom"). Only one Chl *a* peak occurred at six stations in August, including all marine lagoons west of the island of Rügen and two inner stations of the Wismar bay ("summer bloom"). All stations of the Greifswalder Bodden, one station at outer coastal waters close by (no. O133) and one of the stations of the bay of Wismar exhibited three blooms, but with higher Chl *a* concentrations compared to the outer coastal waters ("inner triplet bloom"). The third bloom, however, was not always clearly pronounced at all sampling stations and probably depended on other factors like annual nutrient supply or hydrological flows (see Sect. 18.4). The remaining two groups showed two Chl *a* peaks in April and late August, respectively, but with Chl *a* concentrations at least one order of magnitude higher than at all other stations. Five stations from two highly eutrophic estuarine lagoons showed a delayed off-set of the spring bloom and Chl *a* concentrations of up to 60 µg L^{-1} and represent water bodies characterized by changing in- and outflow events ("advected bloom"). The last group showed the same delayed spring bloom but with Chl *a* concentrations up to two times higher as in the former group ("delayed spring bloom"). All oligohaline estuarine lagoons near the mouths of rivers (Oder and Recknitz) clustered in this group.

18.4 Temporal Variability in Bloom Periodicity

Monthly sampling can result in noisy data over time (Carstensen et al. 2002; Winder and Cloern 2010), that means signals can shift over a longer time period. Therefore, the current data set was Wavelet-transformed (R-package *WaveletComp*, Rösch and Schmidbauer 2018) to filter for recurring phytoplankton periodicities within each coastal water. Detailed descriptions on transformation parameters can be found in Winder and Cloern (2010, and sources cited therein). The wavelet-transformation confirmed the periodicity of the changepoint-analysis.

Coastal waters with three blooms showed either significant recurring periods every 4 months (outer triplet blooms), or three equally significant periods pointing to fluctuations in bloom formation over time (inner triplet blooms). This temporal variability of inner coastal waters with three blooms was possibly related to changing in- and outflow events from several directions. For example, the Greifswalder Bodden represents most sampling stations in the group 'inner triplet blooms', as this lagoon is influenced from the oligotrophic Baltic (east), mesotrophic Strelasund (northwest), and eutrophic Oder estuary (south). Coastal waters with only one major bloom in summer (summer bloom) showed a 6-month periodicity as part of recurrent drop in Chl *a* in March and September (see Fig. 18.2). Inner coastal waters affected by an advected bloom showed unclear blooming patterns over time, which can be related to interannual differences of out- and inflow regimes. Coastal waters of the type "delayed spring bloom" had a strong 12-month periodicity, and only a weak 6-month signal indicating a larger spring compared to autumn bloom. The same was true for less nutrient-affected coastal waters with two blooms (inner duplex blooms) pointing to similar bloom-affecting drivers (see Sect. 18.5). In a meta-analysis of 125 aquatic systems ranging from limnic to marine, blooming periodicities ranged from 4- to 12 months, with 12 month being the most and 6 month the least frequent periods (Winder and Cloern 2010). In the here analyzed data set, 80% of all stations showed at least two blooms per year.

18.5 Driving Factors of Coastal Blooms

Climate variability was identified as a major driver of temporal variability in bloom periodicity. The blooming periodicity shifted in most inner coastal waters during years with precipitation below the annual average, but not at the outer coast (e.g. 2003–2005). Inner coastal waters are highly affected by their catchment area (Schlungbaum et al. 2000; Nedwell et al. 2002). High winter precipitation can reduce Chl *a* concentrations in the following season, whereas precipitation during

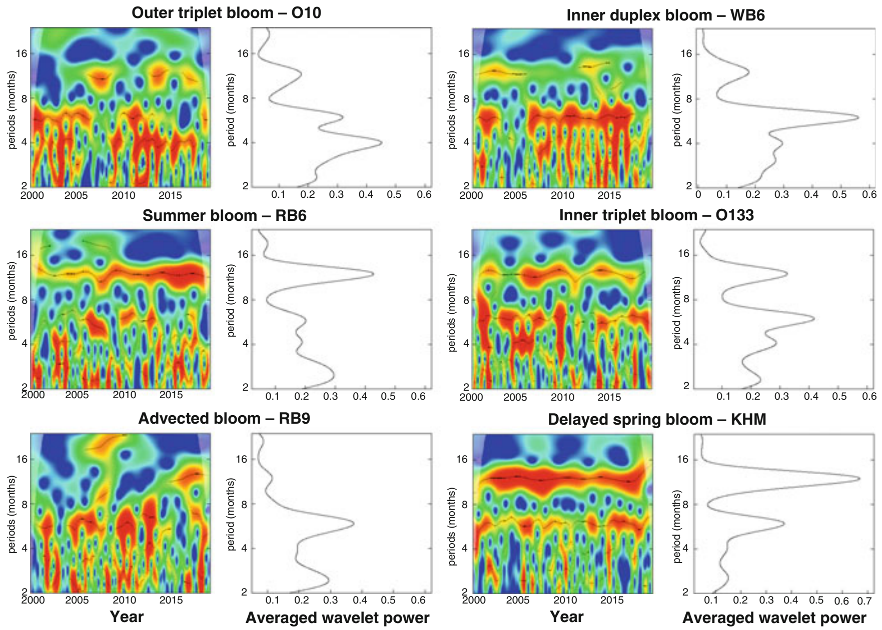

Fig. 18.2 Continuous wavelet power spectra (left side per group) and time-averaged wavelet spectrums of Chlorophyll-concentrations (ln-transformed) for phytoplankton bloom types represent by six exemplary stations of the southern German coast. Red areas indicate periodicities with high intensities, blue areas with low intensities. Peaks in the time-averaged spectrum represent periods explaining most of the temporal variance. Figures were created with the R-package *WaveletComp* (Rösch and Schmidbauer 2018)

summer can increase Chl *a* (Thompson et al. 2015). This close coupling of catchment areas and their adjacent coastal waters is assumed to be the main source for bloom phenotype variability at the southern Baltic Sea coast, and indicates the necessity to reduce the current nutrient supply according to the Baltic Sea Action Plan (HELCOM 2013). It is questionable if the suggested reduction is sufficient, as coastal waters can show hysteretic responses on nutrient declines (Duarte et al. 2015, see Chap. 28, this book). Nonetheless, these blooming patterns can be used as additional classification tool for coastal waters, based not only on their hydrological but also biological phenotype (see also Chap. 10).

The bloom type patterns described above were further assessed using generalized additive mixed modelling (GAMM, *mgcv*-package, function *gamm*, Wood 2011) for all monthly observed variables (Fig. 18.3). The application of this smoothing function revealed that outer coastal waters show three recurring Chl *a*-peaks at days 90, 210 and 300, which already were described as common characteristic in outer coastal waters of the Baltic Sea (Wasmund et al. 2019, see Chap. 16). TN showed no seasonal trends, whereas TP and DIP showed negative trends after the spring bloom, probably related to excessive P-uptake and subsequent sinking of organic material (Wasmund et al. 1998). DIP peaks occurred at days 150 and 250 and dropped immediately before the respective second and third bloom started.

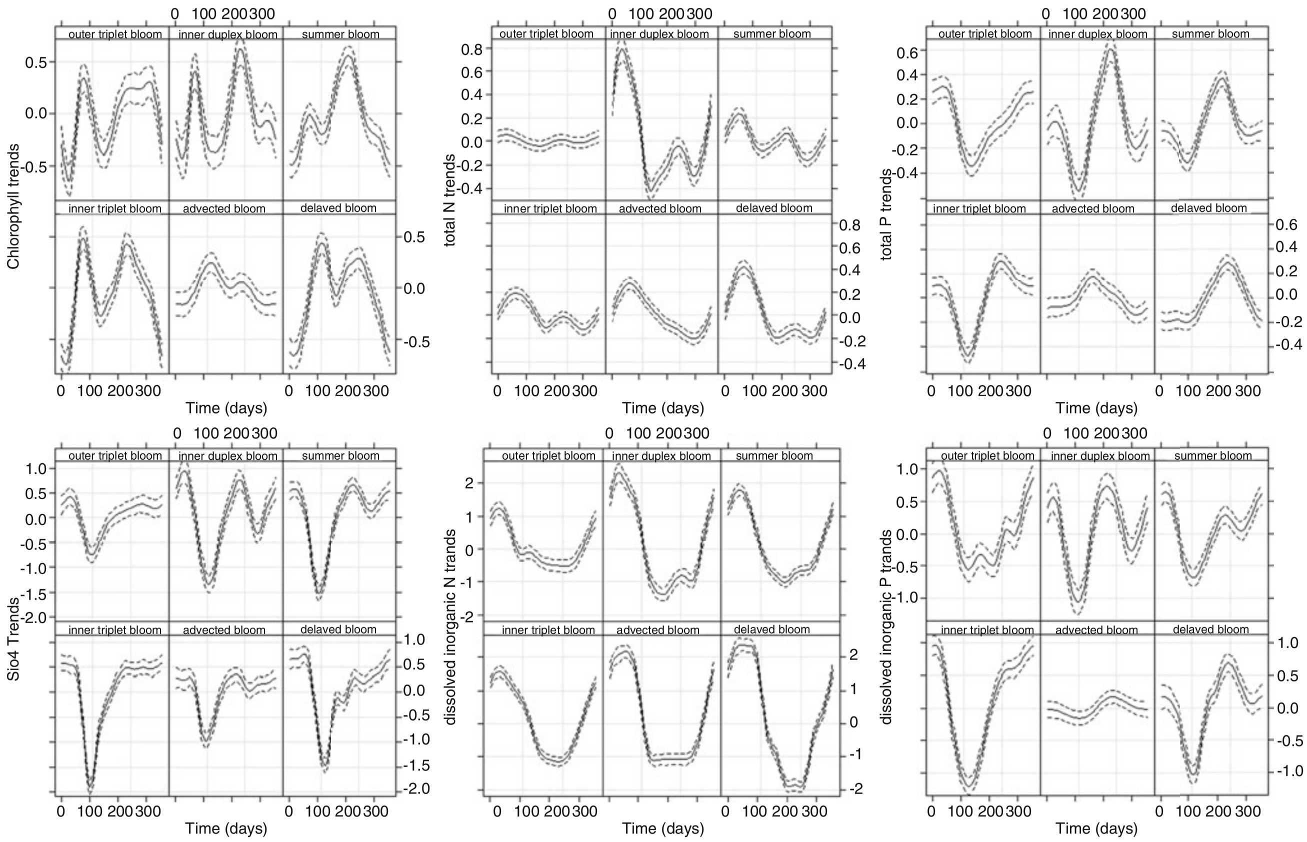

Fig. 18.3 Seasonal smoothed patterns per bloom type obtained by General additive mixed modelling. The x-axis represents "Day of the year" from all grouped stations over the complete observation period, the y-axis the trends for the respective parameter within the year. All parameters were ln-transformed. "Day of the year" was used as smoothing variable, bloom type as grouping variable and a random intercept for the sampling stations (Zuur et al. 2009). "Day of the year" was modelled with a cyclic cubic regression spline (parameter $k = 12$), to force the first and last knot through the same intercept, therefore closing the annual cycle (Wood 2017)

In contrast, DIN decreased steadily reaching its lowest point until the onset of autumn (day 100–270). SiO_4 showed only one negative trend within the year, which can be caused by diatom blooms in this area (Wasmund et al. 2011).

Inner coastal waters with two blooms showed the onset of their blooms at days 90 and 210. TN, TP, DIN, DIP and SiO_4 increased similarly immediately before the blooms, and dropped simultaneously, indicating a possible second diatom bloom (Gasiūnaitė et al. 2005). Interestingly, the GAMM revealed a possible small spring bloom in coastal waters otherwise dominated by a summer bloom. Summer blooms are rather rarely described, and occur usually under cold water conditions at higher latitudes (Sinclair 1978). Seasonal TN fluctuations seemed less pronounced in coastal waters with summer blooms compared to inner coastal waters showing duplex blooms, but TP showed the same trend with a peak at day 210.

Similarly, DIP and SiO_4 decreased twice within a year at days 120 and 290, indicating diatom blooms, while DIN reached its lowest point after 190 days. The water column above the sediments in those coastal waters was always oxygen-saturated (LUNG 2013), allowing only low denitrification rates of 0.5 μmol N m^{-2} h^{-1} (Deutsch et al. 2010), while organic matter decomposition and nutrient remineralization rates may be high (Rieling et al. 2000).

The trend analysis of inner triplet blooms showed that the third bloom may be missing, as already seen in the Wavelet-transformation. Changing periodicities or fluctuations in bloom peak months is a common feature of coastal waters (Cloern and Jassby 2008, see Chap. 17), but GAMM is known to occasionally over-smooth patterns (Binder and Tutz 2006). TN and TP showed increasing trends with peaks during the blooms at days 90 and 250, respectively. SiO_4 showed a single strong negative trend during spring, indicating one diatom bloom. DIN concentrations showed a slower negative trend, reaching a minimum from days 150 to 300. Contrarily, DIP tended to increase after day 120 indicating an imbalance between DIN and DIP availabilities throughout the summer. Inner coastal waters showing summer blooms or triplet blooms differed in Secchi depth and therefore, revealed differences in habitability for submerged vegetation (Blümel et al. 2002; Munkes 2005; Blindow et al. 2016). Stations with dense submerged vegetation can show increased grazing by zooplankton (Meyer et al. 2019) and other filter feeders (see Chap. 13), which may explain low spring phytoplankton biomasses in these areas.

For both bloom peaks, stations with advected blooms showed the lowest Chl *a* trend-increase of all bloom types, indicating a limited self-sustaining potential of phytoplankton. Chl *a* decreased after day 300 (see Fig. 18.1), which coincides with *Marenzelleria* larvae grazing in these estuarine lagoons (Zettler 1996, 1997). These findings indicate the importance of the detritus-based food web in such coastal waters (see Chap. 12). Bioturbating zoobenthos can increase nutrient remineralization and its transport into the water column (Renz and Forster 2014), which can cause a tight benthic-pelagic coupling in these estuarine lagoons. TN and TP showed only negative trends after the spring bloom, while negative SiO_4 trends indicate a small second diatom blooming in autumn. DIN showed a negative trend from days 120 to 300. Interestingly, the second bloom seemed to coincide with a DIP peak at day 180. Aerobic sediment-based mineralization is rapid and can respire 20–90% of total gross pelagic and benthic primary production throughout the year

(Köster et al. 2000). Alternatively, high phosphatase activities can turn over the complete dissolved organic P-pool within an hour (Berthold and Schumann 2020), which may explain these high DIP concentrations.

Coastal waters with delayed spring blooms showed only a slightly negative Chl *a*-trend between spring and summer bloom, pointing to permanently high phytoplankton biomass throughout the growing season. TN concentrations showed two peaks in spring and summer, with the later peak either caused by increased atmospheric N-deposition, N-inflow through the catchment area or N-fixation. Modelled N-depositional rates peak during spring (reduced N-forms) and summer (oxidized N-forms) in the Baltic Sea area (Ruoho-Airola et al. 2012). Precipitation is highest during summer (Berthold et al. 2019) and years with above-average precipitation can increase total N-concentrations up to 60% in these coastal waters (Berthold et al. 2018a). N-fixation is assumed to be of lesser importance, as these coastal waters are strongly light-limited (Schubert et al. 2001). However, an indication for N-fixation is the strong positive trend in TP during summer, indicating an accumulation of P, as either energy storage (Li et al. 2019), or as part of growth-stagnated N-fixating cyanobacteria (Rhee 1974; Hagemann et al. 2019). Furthermore, recent bio-assays found that natural phytoplankton assemblages of these coastal waters were partly dominated by N-fixating species during summer (Berthold and Schumann 2020). SiO_4 showed only one negative trend with a gradual recovery later in the year. Interestingly, DIN concentrations recovered fastest after spring bloom of all analyzed bloom types, indicating a higher supply after the spring bloom and possible onset of N-fixation, as P was still available (Howarth et al. 1988). The DIP trends would support these findings, as DIP showed a strong positive trend even while there was a second Chl *a*-peak. The second DIP peak can also point at higher DIP re-supply rates within these coastal waters, either through grazing (Schiewer 2007), sedimentary remineralization (Berghoff et al. 2000), or elevated hydrological loading from precipitation (Berthold et al. 2019), across the catchment area with its direct and indirect inflows (Berthold et al. 2018a, b).

Overall, the bloom types are mainly a function of nutrient loads, with varying degrees of bloom timing and peaks. The bloom types influence the (re-)appearance of macrophytes, by causing varying degrees of light limitation. As a next step, the parameters investigated here should be coupled with data on solar radiation, water temperature, salinity, light availability, macrophyte cover, filter feeding by zooplankton and macroinvertebrates, sediment re-mineralization rates and enzymatic activity levels (e.g. phosphatase), to create a seasonally resolved phytoplankton model. Such a comprehensive analysis could identify the major limitation triggers in coastal waters, which today still remains an open question.

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Part IV

Combining the Aspects: Ecosystem Service Assessment

The Human Factor: Coastal Social-Ecological Systems

19

Konrad Ott, Martin Benkenstein, Felix Müller, Michael Rauscher, and Hendrik Schubert

Abstract

After the comprehensive description of the ecosystem processes in Chap. 4, the following paragraphs are dedicated to the philosophical, social, and economic aspects of the Baltic human-environmental systems. While in Chap. 2 the foundations of these disciplines have been discussed, those basic ideas are deepened and applied in Chap. 5. The narrative starts with aspects from economics, introducing ecosystem services and approaches to characterize them, mainly from a behavioral science perspective. The next viewpoint is environmental ethics, which provides a reflective and grounding layer for the ecosystem service approach. Thereafter systems-based aspects are applied to human environmental entities in general and the study region as a special case, which is illuminated from a socio-economic viewpoint.

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The last chapters of this book have shown potential contributions of different disciplines for interdisciplinary coastal ecosystem analysis (Chap. 2), the environmental conditions of the Baltic Sea (Chap. 3), the German coast and the terrestrial hinterlands (Chap. 3), and an analysis of the ecological structures and functions of inner coastal water bodies (Chap. 4) and the offshore ecosystems (Chap. 4). All of the respective articles have focused on ecological characteristics and societal conditions, thus the reader might meanwhile have obtained a good impression on the processes and processors of the southern Baltic coastal environment.

In order to move forward to the multiple actual environmental problems of the research area, there will be a stepwise introduction of the important elements and relations of human-environmental systems. Such widening of the scope will be advancing by integrating the viewpoints of economy (Chap. 5.2) and ethics (Chap. 5.3), in order to discuss the ethical suppositions in the ecosystem service approach (Chap. 5.4). Those constituents are integrated in Chap. 5.5 by applying the basic ideas of human-environmental systems approaches.

19.1 Introduction: Human Factors and Normative Analyses

Scientific knowledge cannot tell how humans should behave. It is impossible to derive an "ought"-statement from "is"-statements irrespectively of the number of true "is"-statements. Even if there would be such thing as perfect scientific (biological, ecological, marine) knowledge about the Southern Baltic Sea, political decision makers would clearly stand in need of normative (or: prescriptive) sources of practical knowledge. Therefore, the natural sciences must cooperate with prescriptive disciplines in order to give guidance or make suggestions for policy makers. As argued in Chap. 2, there are three disciplines with prescriptive content: legal scholarship, economics, and ethics. This study abstracts away legal topics. Economics is devoted to efficient allocation of scarce means of production with respect to given human preferences under conditions of trade-offs, risk and uncertainty. The concept of efficiency itself has an ethical meaning, as it is directed against wastefulness. The objective to maximize personal utility or societal welfare (Pigou 2002) is clearly prescriptive. The idea of consumer sovereignty also has some prescriptive force ("preferences are to count"). Both the foundations of economic modeling and legal policies are to be reflected in economic theory, political philosophy, and in ethics. This remains true, if economics, law, and ethics are applied to environmental topics. Although the methods and the conceptual frames of economics, legal scholarship, and ethics are different, they should be regarded as an interconnected cluster of normativity within the natural sciences.

Table 19.1 Different definitions of the term "Ecosystem Services"

Daily (1997)	Ecosystem services are the *conditions and processes* through which natural ecosystems, and the species that make them up, sustain, and fulfill human life
Costanza et al. (1997)	Ecosystem *goods* (such as food) and *services* (such as waste assimilation) represent the benefits human populations derive, directly or indirectly, from ecosystem functions
Boyd and Banzhaf (2007)	*(Final) Ecosystem services* are components of nature, directly enjoyed, consumed, or used to yield human well-being
Fisher and Turner (2008)	Ecosystem services are the aspects of ecosystems *utilized (actively or passively)* to produce human well-being
Millennium ecosystem assessment	– Ecosystem services are the benefits people derive from ecosystems – Ecosystem services are the benefits people obtain from ecosystems and *also the processes that produce* or support the production of ecosystem goods
TEEB (2010)	Ecosystem Services are *the direct and indirect contributions* of ecosystems to human well-being. The concept "ecosystem goods and services" is synonymous with ecosystem services
Haines-Young and Potschin (2010)	Ecosystem services are the contribution which the biotic and abiotic components of ecosystems jointly and directly make to human well-being, *an "end-product" of nature*
Burkhard et al. (2012a)	Ecosystem services are the contributions of ecosystem structure and function—*in combination with other inputs*—to human well-being

19.2 Economic Aspects of Human–Environmental Relations

Already Westman (1977) used the headline "How Much Are Nature's Services worth?" for a paper published in Science. The term "Ecosystem Services"[1] (ESS) occurred first 1981 (Ehrlich and Ehrlich 1981). However, the idea, that ecological systems are beneficial for human beings and provide contributions for their well-being is much older in academic discussions.

Taking this into account, we would like to look back on ecosystem service research and its special contributions on coastal ecosystem research. And we would like to go one step further and discuss coastal ecosystem service research from a behavioral science perspective.

19.2.1 Starting Points in Environmental Economics

"Ecosystem Services (ES) are the ecological characteristics, functions, or processes that directly or indirectly contribute to human wellbeing: that is, the benefits that people derive from functioning ecosystems" (MEA 2005, see also Table 19.1). But

[1] In several instances of the forthcoming texts, the term "Ecosystem Service" is shortened to the abbreviations ESS or ES.

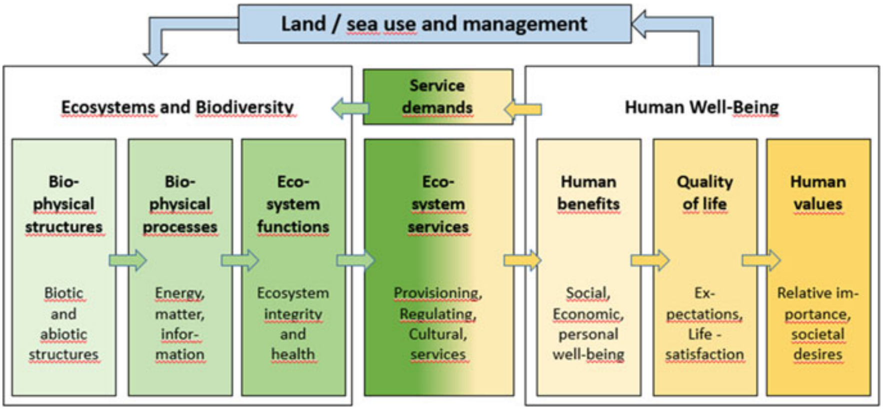

Fig. 19.1 Cascade model: Ecosystem service components from structure to functions to services to benefits to value (after Potschin and Haines-Young 2017)

how do these connections and interrelations work in specific local settings in detail? This is one of the central questions of ES research, which was intensively discussed during the past decades.

The first field of ES research that is relevant for our Southern Baltic coastal ES analysis is the discussion about the interactions between the ecological environment and the social and economic system. A result of this discussion is the “cascade” model by Potschin and Haines-Young (2017) visualized in Fig. 19.1. We can see that the chart starts with bio-physical structures or processes, goes on via functions to services and then—leaving the ecological environment and entering the social and economic system—produces benefits to human beings and creates value for single persons or for parts of society (Costanza 2008).

This cascade model was heavily discussed and extended. Especially the extension by the dynamic system model introduced by Costanza et al. (2017) is relevant for our research because—within the social and economic system—constructs were integrated that are able to explain why the services from ecosystems are able to create benefits and value. These constructs are, for example, images, needs, and preferences. We will discuss these constructs and their relevance to explain the interrelations between services and benefits or value later on in this chapter.

The second field of research relevant for us is the discussion about ecosystem service classification systems. A meta-classification was already introduced with the cascade model by MEA (2005). The authors discriminate between provisioning, regulating, cultural, and—sometimes—supporting services (Hernandez-Blanco and Costanza 2019):

- *Provisioning services*: The ecosystem provides human beings with goods such as water, timber, or food.
- *Regulating services*: The ecosystem creates value by regulations of ecosystem processes such as flood control, water purification, or climate control.
- *Cultural services*: The ecosystem creates non-material benefits such as spiritual, recreational, or aesthetic benefits.

- *Supporting services*: The ecosystem provides structures and processes that create value indirectly because they are necessary for the three other types of services. These services are also comprehended as attributes of "ecosystem integrity." The key components are ecosystem structures and ecosystem processes (Müller 2005). In order to avoid wrong accounting results, this class has been neglected in most recent classification systems for ecosystem services.

Within these categories, a lot of research was done to create classification systems for ES in general or for special regional ES, e.g., for coastal ES (Sukhdev and Kumar 2008; Böhnke-Henrichs et al. 2013; Kandziora et al. 2013; US EPA 2015; La Notte et al. 2017; Haines-Young and Potschin-Young 2018). One of the most prominent assessment concepts are so-called ES matrices conceptualized, e.g., by Burkhard et al. (2009, 2012b), Fürst et al. (2009), and Koschke et al. (2012). Through these matrices, the potential of ecosystems to supply services to human beings is calculated. Therefore, on the vertical side of the matrix ecosystem structures and processes that represent ecosystem integrity are classified. Moreover, on the horizontal side provisioning, regulating, and cultural services are categorized. In this way, experts are able to estimate for each cell of the matrix the potential of supporting services to create provisioning, regulating, or cultural services. Figure 19.2 demonstrates the layout of such an ecosystem service matrix. Within this ecosystem service matrix-structure a lot of research was done to create matrices for special regional ES, e.g., for coastal ES (e.g., Burkhard et al. 2014; Stoll et al. 2015; Müller et al. 2020; Schumacher et al. 2022). By using the ecosystem service matrix we are able to aggregate ecosystem structures and processes into land use classes and translate them into ecosystem service potentials, respectively, ecosystem service offerings. In the following, we try to answer the question how we can transform these offerings into economic value.

19.2.2 The Social-Economic and Behavioral Science Perspective

From the social-economic perspective, we have to ask how the services provided by ecosystems create value for human beings. This question is not trivial: between service offerings on the supply side and value creation on the demand side, there are a lot more than the benefit-construct from the cascade model. Behavioral science research has identified many intervening variables and constructs. These variables and constructs are responsible for the transformation of sole offerings like fish or landscape into valuable offerings like food or holidays. Variables and constructs that are responsible for this transformation are, for example, images, needs, and preferences. They "translate" sole offerings into valuable services and benefits. Let us look how this transition works.

When we try to answer the question how benefits and value occur, we have to look at research streams coming from behavioral sciences. Especially consumer- and buying behavior-research analyze why customers buy and use products and services

		Regulating Services				Provisioning Services				Cultural Services			
		Climate Regulation	Water Flow Regulation	Erosion Regulation	...	Fish	Timber	Aquaculture	...	Tourism	Cultural Heritage	Landscape Aesthetics	...
Ecosystem Structures and Processes	Sediment												
	Reaf												
	Sand Bank												
	Dike												
	Beach												
	Salt Marshes												
	Forest												
	...												

Fig. 19.2 Exemplary ecosystem service matrix: from ecosystem structures and processes via ecosystem types to ecosystem services

to create personal benefits. This research postulates—similar to ES research—system models of different drivers of benefits and value.

Figure 19.3 shows such a systems model. Starting in the left side ecosystems and their service offerings create stimuli for the potential user of that services. The stimuli initiate information processing and lead to problem recognition. At that state the decision-making process starts and ends with the decision to use or not to use the ecosystem service offerings. Moreover, the usage leads to value and satisfaction or dissatisfaction. Within the decision process during the early phases, the potential ecosystem service user takes a lot of decision-making and general motivation variables into account.

As soon as we want to interconnect the ecosystem service offerings and the value creation on the demand side, we have to look on these decision-making and general motivation variables. In the following, we will use motivation as such a variable.

One of the most relevant variables for the transition of offerings into benefits and value is motivation, because motivation creates behavior (Sheth et al. 1999). We distinguish between emotional and cognitive processes within the motivation variable. The emotional processes stimulate a behavioral response, while the cognitive process provides specific directions to that response. The terms motivation and need are often used interchangeably. The best-known and most powerful categorization is

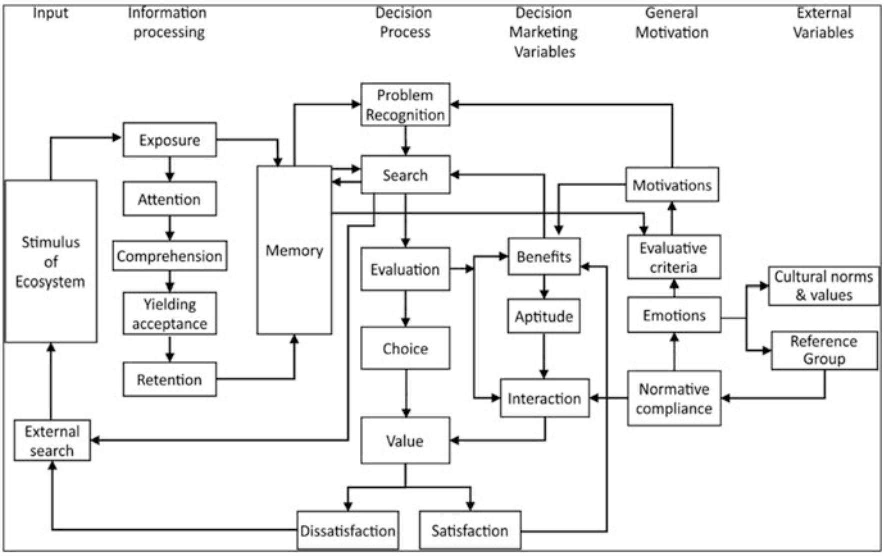

Fig. 19.3 Model of ecosystem value creation from a behavioral science perspective, following (Engel et al. 1978)

Maslow's need hierarchy (Maslow 1970), a macro theory designed to account for most human behavior in general terms (see Chap. 2).

19.2.3 Integrating ES-Research and the Behavioral Science Perspective

Maslow's theory helps to explain human behavior and therefore is able to link ecosystem service offers to benefits and value because only those offerings are valuable for human being that fulfill special needs in Maslow's pyramid and satisfy selected motivations. Therefore, we integrate Maslow's Pyramid of needs into the ecosystem service matrix. The result is shown in Fig. 19.4.

Using this extended ecosystem service matrix, we are able to translate ecosystem processes and structures into ecosystem service offerings. Furthermore, we are able to calculate to which extent these offerings are able to satisfy human needs and motivations.

When we try to assess the transformation of ecosystem service offerings into need-satisfaction, we have to take into account that the same ecosystem service offering is able to contribute to different levels of needs in Maslow's pyramid. For example, fish can—of course—fulfill physiological needs when people are hungry. But fish can also contribute to esteem and even to self-actualization if a fly fisher is able to catch a big trout.

On the other hand, an ecosystem service offer can also stimulate conflicting motivations. Let us take eel as an example. Eel is able to fulfill physiological

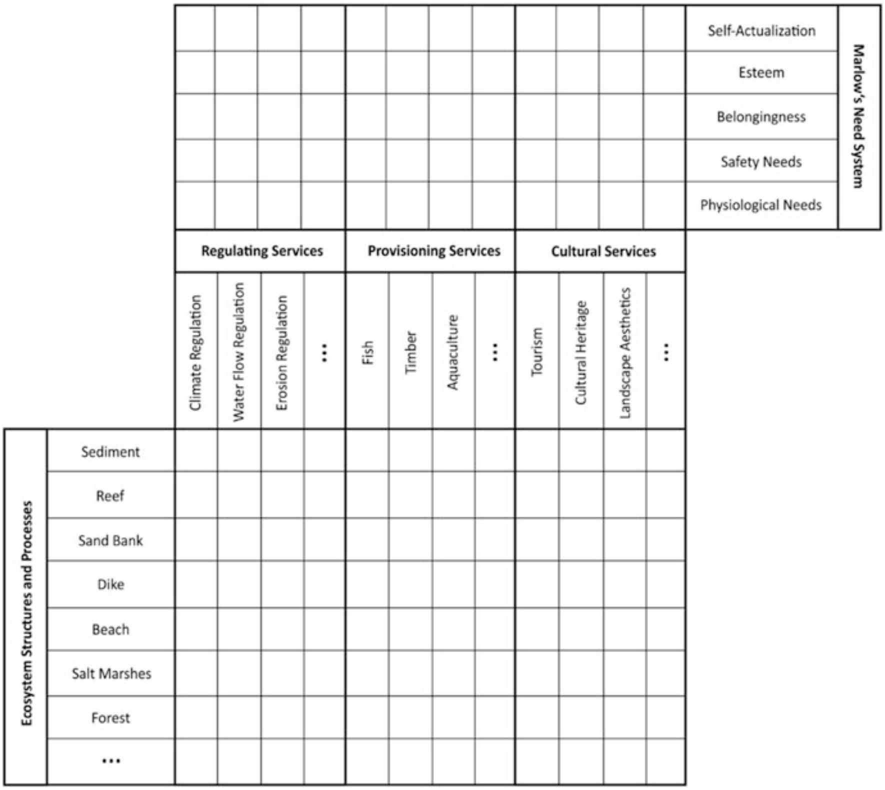

Fig. 19.4 The potential structure of an extended ecosystem service matrix

needs again when people are hungry. However, eel is a threatened species. Therefore, the desire for self-fulfillment may create a motivation not to eat the eel. These conflicts are typical in the system of motivations.

19.3 Environmental Ethics: Patterns of Reasoning

Humans perform cognitive operations as they schematize, classify, grade, judge, and type. They also do so with respect to evaluations. This subchapter argues that all conceptual schemes used for environmental evaluation are, finally, to be grounded in the universe of environmental ethical discourse (for a philosophy of environmental evaluations see Ott 2020, Ott and Reinmuth 2021 with further literature). This claim holds true for ES as well. Therefore, ecosystem service assessments should incorporate a reflective ethical layer of inquiry which is outlined in this subchapter.

Table 19.1 gives an overview on "ecosystem service" definitions. At the core of all ESA definitions are the common idea to bridge the gap between the performance of ecosystems and human welfare. This idea is expressed by the metaphor of a “cascade.” Trivial to state, that valuable benefits which humans obtain from ecosystems contribute to human welfare. Disservices diminishing welfare are abstracted away in the definitions and are not addressed here. Definitions are highly generic, and specific ESA studies must specify them with respect to geographical locations, ecologies, human stakeholders, cultural values, and trade-offs.

A terminological caveat is at place here: Author dislikes the term “service,” because it stems from economics of service industries and it may transport misleading connotations. Falling prey to this “service” terminology, we may end up perceiving nature not in its ecological naturalness and its fertility, resilience, diversity, and richness, but in analogy to service industries (as pizza service, laundry, etc.). To avoid this pitfall, we should use the established term “service” as technical term referring to the many modes by which nature can be beneficial to humans. Any service has, by definition, a positive value to humans (individuals, groups, stakeholders, communities, etc.).

If one reflects service categories within ESA, one necessarily touches ethical topics. Therefore, this subchapter wishes to make the significances of such ethical reflections for ESA more explicit than this is usually being done in the literature.

Ethics rests on some basic concepts. A basic distinction is between moral and ethics. Morals is a network of emotions, intuitions, convictions, and believes that forms the characters of moral persons (= agents). Morals strongly vary across cultures and history. Ethics is reflective about moral believe systems. Ethics can be distinguished in different layers of inquiry:

1. Deontic logic and analysis of moral concepts
2. Metaethics: meaning of moral discourse
3. Ethics of “Good Life” (Aristotle: eudaimonia)
4. Normative Ethics (as Kantianism, Utilitarianism, Discourse Ethics)
5. Applied Ethics (= Practical Philosophy in Fields of Human Practices)
6. Case Studies

Environmental ethics is one field of applied ethics (layer no. 5), presupposing a set of assumptions from layers 1–4. This chapter only addresses layer no. 5, for general ethical theory, see Ott (2005a), Tugendhat (1994) and contributions in Brune et al. (2017). Ethics of good (meaningful, flourishing) human life (level 3) constitutes a reflective background for concepts of welfare, benefits, and utility being present in ESA. Normative ethics constitutes a reflective background for ideas about commitments, obligations, equity, and fairness being useful for analyzing trade-offs and conflicts about production, maintenance, and distribution of ecosystem services. It makes a difference whether agents appreciate ecosystem services (value) or whether agents are committed to maintain it (obligation). ESA focusses benefits and values but rather unspecific with regard to obligations, justice, laws, and commitments.

On level 5, one can distinguish three approaches in environmental ethics (Ott and Reinmuth 2021). The “classical” approach tries to resolve the problem of inherent moral value in nature (demarcation problem, Ott 2008, Sober 1995) and it derives from such resolution a concept of nature conservation (Taylor 1986). This approach wishes to overcome anthropocentrism. Since ESA is an anthropocentric concept, inherent value approaches should not be ignored because the demarcation problem is often mentioned by participants of ESA studies and by students in ESA-courses. Thus, ESA should conceive its relation to the demarcation problem. It can do so by ignorance, denial, or abstraction. Environmental ethics dislikes ignorance and denial because the demarcation problem is essential for the entire discipline (Attfield 2014; Warren 1997; Krebs 1999; Sober 1995). If the demarcation problem is abstracted away for the sake of identification and measurement of ecosystem services, it remains an "open question" for ESA-scholars. It becomes a salient point for ESA being located outside of ESA. If so, it deserves attention.

A second approach is post-modern environmental ethics. In post-modern approaches, narratives, literature, pictorial representations, alternative media approaches, and criticism against Western economic rationalities play an important role (Morton 2016, Haraway 2016 for criticism see Ott 2019). Post-modern approaches take no interest in ESA (Haraway 2016) but rather regard ESA as a repugnant neo-liberal commodification of nature. Post-modern approaches are left aside.

A third approach is environmental pragmatism (Norton 2005, 2015). Environmental pragmatism exists within the tradition of philosophical pragmatism (cf. Schneider 1963, Chapters VIII and IX; Minteer 2006), as it takes its starting point in different kinds of established human practices in dealing with nature (agriculture, forestry, sailing, hiking, gardening, etc.) and explicates the many different values being involved in such practices. Environmental pragmatism aims, in a reforming and civic way, to make such practices more compatible with sustainable preservation of nature (see Sagoff 1988 for a reconciliation of environmentalism and political liberalism). Environmental pragmatism can adopt ESA without reservations. Norton (2015, pp. 179–198) argues that pragmatism explores a toolbox of schemes for environmental evaluation. Norton distinguishes conceptual-analytic and behavioral-action tools. ESA belongs to the former tools. In this subchapter, a theoretical approach is endorsed which combines the traditions of continental philosophy of discourse (Hönigswald 1937; Apel 1976; Habermas 1981) with environmental pragmatism. ESA can and should be embedded within this theoretical approach. If so, ESA is seen as a helpful tool (= device) for deliberation and reasonable choice (Hiedanpää and Bromley 2002) in environmental policy-making underpinned by a reflective ethical layer and embedded in the theoretical paradigm of discourse-oriented environmental pragmatism.

Within the general approach, six specific lines of reasoning constitute a universe of environmental ethical discourse that underlies the ESA toolbox. These six lines of reasoning go ethically beyond sociological ESA studies as they rather provide reasons why ecosystem services *should* be appreciated, preserved, and restored (O'Neil et al. 2007). They substantiate the factual appreciation of ecosystem services ethically (Jax et al. 2013).

19.3.1 Dependence and Reliance

Supporting and provisioning ecosystem services are, ultimately, grounded in reliance-arguments. Arguments from dependence and reliance claim that human beings, as embodied and precarious beings, are dependent on a continuous metabolism with nature, the maintenance of which requires a careful use of natural resources and environmental media. Reliance is common, but differentiated. Humans *should* care for nature and ecosystem services either out of prudence or out of duties against other persons being reliant on ecosystem services. Maintenance of ecosystem services can be in the prudent interest of a societal unit (household, community, state) or there can be moral obligations not to impair ecosystem services upon which other people are reliant. To impair and destroy ecosystem services needed for decent livelihoods count as ecological victimization from a justice perspective (Martinez-Alier 2002).

Supporting services point to basic structures and functions of ecological systems which make human-ecological systems productive and resilient (see Nielsen et al. 2019 for theory, Meyer 1997 for ecosystem function). Even if economists discard supporting ecosystem services from ESA studies wishing to avoid economic double-counting, reliance on life-supporting ecosystems, as fertile soils, forests, rivers, ocean waters, groundwater tables, pollination, etc. is beyond doubt. From an environmental ethics perspective, supporting services are close to so-called systemic values, as fertility (Rolston 1988, 1994, 1999). Ecosystems have systemic vital value which are of non-moral goodness (Rolston 1999, pp. 43, 360). The wording "supporting" may even underrate such basic ecosystem services. Supporting services point out that "something is at work" within ecological systems without which there will not be other ecosystem services. Supporting services make provisioning, regulating, and cultural services become possible and real.

Nature also provides specific resources for meeting basic human needs (water, food, shelter, overview in Dudley 2011, Chap. 5). Provisioning services are, however, mostly mediated by human labor: freshwater, cereals, fish, beers, etc. Therefore, ESA studies must conceive human-ecological land use systems in close correlation in order to understand provisioning services. Environmental pragmatism points to the economic side of provisioning services, as agriculture, forestry, fisheries, grazing systems. To pragmatism, the approach to provisions is Lockean: Nature must be mixed with human labor because wild nature as such contributes roughly 1% of utility to humans while cultivated systems contribute 99% (Locke 2002, p. 19). Provisioning services become manifest in yields which can be processed further to food, textiles, furniture, etc.

19.3.2 Eudaimonic Values

Arguments from cultural or eudaimonic values ("eudaimonia" = good life) claim that experiences of nature are an essential part of a rich, successful and meaningful life (Ott 2016; Chan et al. 2016; Holland 2006; Hargrove 1992). Eudaimonic values have large overlap with cultural ecosystem services. From an environmental ethics perspective, cultural ecosystem services are not just a speculative add-on to measurable provisioning and regulating services but are essential to the overall ESA approach (Ott and Reinmuth 2021). ESA studies should not underrate them. Ethics can shed light into the deep background of eudaimonic values (Firth 2008; Benton 2008) since they are not simply "naturally given" but mediate between natural phenomena and cultural traditions (Ott 2016). While ESA can remain at the level of preference satisfaction, eudaimonic values refer to the idea of a worthwhile life with and within nature (Holland 2006).

Coastal zones are paradigm locations for eudaimonic values. The matter of fact that many humans migrate to coastal zones and coastal zones are prominent tourist destinations count as strong sociological evidence. Eudaimonic values of coastal life are, however, ambivalent with respect to nature conservation because they may imply over-tourism and, in economic parlance, provoke congestion effects. Authorities have to find a delicate balance between open access, regulation, and protected areas. This seems to be true for the Baltic. Therefore, we should take a closer look on eudaimonic values grounding cultural ecosystem services.

Eudaimonic values are divided into different ways of enjoying nature, such as promoting physical and mental health (Dudley 2011, p. 104), experiences of natural beauty (Seel 1991, Saito 2014, Sepänmaa 2014), a sense of being at home (Scruton 2012) and spiritual recuperation in nature. Eudaimonic values explain why many people are unwilling to forego contact with nature in their lives. Norton (1988) argues that experiences of nature often have a transformative effect on their attitudes toward life ("transformative values"). These transformative values point to environmental virtue ethics (see below). The same applies to the view that nature is an indispensable "sphere of resonance" for human experiences (Rosa 2014). Eudaimonic values explain why landscapes can be "therapeutic" (Gesler 1992).

The commonly shared value of human health leads to the question whether and to what extent specific natural sites (forests, coastlines, mountains) are beneficial to physical and even mental health. Healing, refreshing, and recreational effects of forests and coasts and the health-promoting activities of hiking and bathing are not denied from scientific medical points of view anymore. Since medical research gives salient focus on the neuro-immune system (Hyland 2011), new connections between natural environments, human outdoor activities, and maintenance and recovery of health might be established. At the University of Exeter (Prof. Lora Fleming), there is a center of research investigating specific health-related topics in coastal populations (life expectancy, mental disorders, suicidal rates, strokes, etc.). With some caveats in mind, there are reasons to believe that coastal populations are, on the average, in a better health condition. Bell et al. (2015) see coastal zones as paradigms of therapeutic landscapes.

Such community medicine perspective (Fleming et al. 2014) should be augmented by cultural studies since health-related and cultural motives intertwine in environmental movements as in earlier times, for example, in the lifestyle reform movement or the German "Wandervogel" movement (Wolschke-Bulmahn 1990; Wedemeyer-Kolwe 2017). Nudism became prominent in Germany since 1900 since it could point to the presumed healthiness of being naked in the outdoors, especially on the beach (see Andritzky and Rautenberg 1989). Coastal zones became prominent locations for nudism in the German Democratic Republic also. As the example of nudism shows, concepts of bodily health are always mediated with cultural ideas about a flourishing human life. A historical-cultural investigation on the origins of tourism and recreation at the Baltic coast came to the result that health-related ideas played an important role in emerging tourism since the nineteenth century (see Chap. 6.3). This has not changed since then. At present, health care is of high cultural significance in all societies surrounding the Baltic Sea. This significance has increased since the Covid-19-pandemic (Popp and Ott 2020). Health effects of therapeutic landscapes can, in principle, be addressed by economic methods, as payments for wellness locations and travel cost analysis.

Some other cultural services, however, remain obscure and opaque to scientific and economic methods. This seems to be true for, e.g., “beauty” and “spiritual encounters with nature” (see contributions in Bergmann et al. 2013).

Generally, ESA-studies should be warned against underrating cultural services that can neither be perfectly monetized nor measured in physical terms (“How many tons of beauty?”). Many scholars fill the gaps of ESS approach with ideas of participation, stakeholder involvement, and deliberate decision-making. Therefore, cultural services are another reason to perform transdisciplinary studies (Chap. 2). With respect to deeper layers of cultural services, other approaches in the humanities, as cultural history, history of landscape painting, history of nature conservation, cultural anthropology, and religious studies (see contributions in Kearns and Keller 2007, Jenkins et al. 2017) can contribute to an in-depth understanding of cultural services, especially spiritual ones. Phenomenology of nature investigates how cultural ecosystem services reveal into mental states (Böhme 1997; Abram 2004). Understanding spiritual services must go beyond ecosystem analysis. Phenomenological expressions of how atmospheres, auras, and sacred sites are perceived and how they constitute specific mental moods may come close to aesthetic, transformative, and spiritual encounters with nature. It is fair to say that cultural ecosystem services must go far beyond economic assessment, as in contingent-valuation studies. If some persons become attuned to special places and sacred sites (see contributions in Mallarach 2012), their willingness to accept compensation for losing such sites may drop to zero.

19.3.3 Intergenerational Responsibility

The values of the first two categories of values (reliance, eudaimonic values) can and should be prolonged into an intergenerational perspective. Long-term policies for safeguarding ecosystem services must suppose some intergenerational obligations (see Düwell et al. 2018). From an ethics perspective, there must be a rationale why current generations are not entitled to consume the sources of ecosystem services away within their lifespan but should bequeath a fair intergenerational legacy in terms of ecosystem services. The ESA approach as such does not entail such rationale even if long-term thinking might be implicitly supposed. Such fair ecological legacy should be grounded in an egalitarian standard of intergenerational equity prescribing that average members of future generations should be equipped with as least as much ecosystem services as present generations (Ott 2005b). Under such obligation, environmental evaluation becomes a matter of the prudent art of long-term thinking (Klauer et al. 2013). Ecosystem services become an asset within such legacy, which may shrink or enhance within the chain of generations.

The topic of fair legacies leads to concepts of *sustainability* (see Ott and Döring 2011). As it has been argued elsewhere, there are reasons to adopt the concept of strong sustainability (Ott 2009; Daly 1996). Within the concept of "strong" sustainability and its constant natural capital rule, nature conservation represents an essential dimension of sustainability policies (Ott 2015a, b). Strong sustainability also entails a restoration rule: If the stocks and funds of natural capitals from which ecosystem services flow have been diminished in the past, societies should invest in natural capitals by means of restoration ecology (Zerbe and Ott 2021). This rule demands to increase the flows of all kind of ecosystem services because cultural ecosystem services cannot be substituted by provisioning services (and vice versa). If an egalitarian standard is taken seriously, a fair intertemporal legacy must include all kinds of ecosystem services undiminished. If so, it would be unfair if present generations maximize provisioning services at the expense of cultural services in the future.

Since provisioning services will be crucial for meeting basic needs of future generations, the famous WECD (1987) definition of sustainable development ("(. . .) meeting the needs of the presence without compromising the ability of future generations to mee their own needs") focusses conservation of supporting and provisioning services. The WCED definition is silent on cultural and regulating services.

If there are strong reasons to increase regulating services in order to combat climate change in the twenty-first century, this might be appropriate from an intertemporal perspective. The urgency to enhance food security for a growing population and the urgency to produce negative emissions in order to keep climate change likeliness, endanger cultural ecosystem services. ESA scholars should, on reflection, keep this trade-off in mind.

Strong sustainability has been applied to coastal zones via an interpretation of SDG 14 ("Life below water") by Neumann et al. (2017). Since coastal zones provide all types of ecosystem services, unspoiled or restored coastal zones are a high-rank legacy to future generations. Depending on the definition, coastal zones reach out for many miles in the hinterland, covering many ecosystems and landscapes. Therefore, coastal zones are precious assets in the overall stock of natural capitals contributing to the sustainable wealth of a country. Coastal zones have to be defended against the imperatives of tourism, shipping routes, harbors, and even offshore-wind farms. Marine and coastal spatial planning seems mandatory for long-term sustainable development (SRU 2004).

19.3.4 Environmental Virtue Ethics and Biophilia

The values and commitments of these first three categories (reliance, eudaimonistic values, and fair intergenerational legacy) lead inevitably to the question of what kind of person one wants to be in the worrisome times of the Anthropocene. This question concerns different attitudes toward nature, including one's own biological-embodied, aging and mortal nature. Such line of reasoning leads to the realm of environmental virtue ethics (see contributions in Cafaro and Sandler 2005).

Preservation and care, curiosity, attentiveness, restraint, protection, consideration, moderation, simplicity, but also joyful devotion, humility, affirmation of life and gratitude are some of the relevant attitudes within environmental virtue ethics, but also vices as gluttony, arrogance, and greed (Cafaro 2004). It is open for further research whether there are specific virtues related to the sea, as sobriety or tranquillity of mind, but also courage. Which attitudes might be implied in the parlance that one "loves" the sea? Interesting enough, terrestrial beings, as humans are, can "fall in love" with the alien world of the sea.

Attitudes and virtues are crucial since cultural ecosystem services are composed of emotions (see Kals et al. 2000), perceptions, traditions, longings, habits, and attitudes. Environmental virtue ethics constitutes a background of relevance for weighing trade-offs between different kinds of ecosystem services. Depending on their virtues and vices person may prioritize some services at the expense of others. Virtue ethics is not directly addressed by ESA studies but belongs to the background of environmental evaluations.

Environmental virtue ethics also is of relevance to moral and environmental education. Thus, environmental virtue ethics strongly supports the idea to educate children and young adults in terms of ecological literacy in general and ocean literacy in particular. Ocean literacy would be incomplete without philosophy (Scholtz 2016) and ocean ethics (Dallmayer 2003).

Eudaimonic values and environmental virtues may have deep roots in evolutionary anthropology. As a legacy of many millennia of co-evolution, human beings may possess a biophilic inclination structure (Wilson 1984; Kellert 1997). The concept of biophilia means a profound disposition in the human mind to affiliate with living beings and living (or lifelike) processes. "Affiliation" means to have close contact. The human mind has formed by interaction and interference with nature, which clearly included foraging, and hunting but also knowledge of animals and plants, symbols and imageries (Levy-Strauss 1981).

Biophilic inclinations can serve as anthropological and evolutionary explanation why ecosystem services, including cultural and even spiritual ones, are appreciated across cultures even if there are many cultural differences as well (see contributions in Ehlers and Gethmann 2003). Reference to the biophilia-hypothesis can explain why ESA can, in principle, be applied globally even if non-Western cultures may not be familiar with the Western parlance of "service" (or may dislike them).

A comprehensive typology of biophilic values is given by Kellert (1997). Levy (2003) presents a fine-grained analysis of the biophilia-hypothesis. Levy (2003, p. 246) concludes that humans "benefit from contact with a non-human world in ways that are reasonably called 'aesthetic' and 'spiritual'". If so, biophilic inclinations reveal itself in eudaimonistic values and environmental virtues.

Many biophilic ways of life are practical ones. Bird watching, diving, hiking, gardening, musing with pets, even going by bicycle through open landscapes are instances of modern biophilic practices. The opposition to biophilia is retirement from nature and a devotion to machines, money, factories and offices, television, etc. Fromm (1974, 1976) construed an opposition between biophilia and the virtuous attitude of "being," on the one hand, and necrophilia and the vicious attitude of "having," on the other hand. This is of relevance for ESA since one can adopt the attitude to possess the sources of ecosystem services or enjoy them with a willingness to share them with others.

19.3.5 Religion and Spiritual Services

The term "spiritual service" is uncommon to the field of religious studies (Jenkins et al. 2017), but it may serve as a purely technical term for the multitude of perception and experience that touch the sphere of the sacred within nature. From an ESA-perspective, spiritual services encompass all spiritual ecosophies and worldviews (as "pacha," Vedic wisdom, Daoism, "obuntu," Deep Ecology, etc.). Without reference to specific religious traditions, the category of spiritual values remains abstract. Any religion is a specific one. An overview of sacred sites and spiritual attitudes is given in Ramakrishnan et al. (1998) and Mallarach (2012). Environmental theologies in the spirit of the Hebrew Bible are given in Hardmeier and Ott (2015) and Vogt (2021) via a correction of the misreading of Genesis 1 as simply "subduing" nature.

In a broad sense of spirituality, also Romantic traditions may count as spiritual ones. The Romantics saw nature as "wonderland" full of bliss lifting the spirit to a "great secret." Out-reaching in this respect was Friedrich Hölderlin whose poetry reveals a spiritual reverence for nature (Mögel 1994). Romantic encounters with nature start with intense aesthetic experiences with nature, but it moves beyond beauty because aesthetic experience seems to reveal something being "more" than just beautiful (Ott 2013). As we know from the history of Romanticism, the Baltic Sea was the paradigm location of the mysteries of Northern latitudes. The paintings of Caspar David Friedrich reveal such locations. The category of cultural services should not just refer to the mundane practices of current mass tourism at the Baltic coastlines (recreation, beauty) but should keep such spiritual traditions in mind.

As Cooper et al. (2016) argue the ESA has conceptual and methodical problems to incorporate spiritual services properly. It transcends the scope of economic techniques (as contingent-valuation, willingness to pay, willingness to accept, travel costs). In a secular culture, many persons may be reluctant to talk about spiritual experiences in interviews, via questionnaire, or in public settings. Spiritual ecosystem services are a paradox for ESA: On the one hand, it must make room for spiritual services from within the ESA tool box, because they matter much to many people all around the world (contributions Jenkins et al. 2017), while, one the other hand, it wishes to abstract away such obscure values for methodological reasons. Environmental ethics argues that this paradox should not be resolved in a way that saves the method but eliminates the spiritual dimension from ESA.

19.3.6 Inherent Moral Value

The category of *inherent (=intrinsic) moral values* points beyond anthropocentrism. Since ESA is anthropocentric, the category of inherent moral value is abstracted away. From an environmental ethics perspective, ESA should be aware of such abstraction and, moreover, should be able to say a word about inherent moral value if participants of ESA studies claim that some natural beings should be protected for their own sake and not just for the sake of services they bring about. If this category of inherent moral value is applied to specific entities, it implies respect and protection for their own sake. The idea of overcoming anthropocentrism was at the heart of environmental ethics since its origins (Routley and Routley 1979; Callicott 1980). Different non-anthropocentric solutions of the demarcation problem are subsumed in the category of "physiocentrism." Even anthropocentric approaches must give due consideration to the demarcation problem after they have harbored eudaimonic and spiritual values, intertemporal responsibility, virtues, and biophilic attitudes.

In physiocentrism, different criteria of direct moral consideration are discussed (e.g., sentience, perceptive awareness, being alive) and claimed as morally relevant characteristics or criteria.

An appropriate solution to the demarcation problem should combine the two characteristics of sentience and the ability to communicate into a gradable concept of openness to a species-specific environmental world (*Weltoffenheit*) (Ott 2015b), followed by a complex casuistry that ranges from chimpanzees and whales to fish, jellyfish, dragonflies and spiders, for some authors even to plants since plant can exchange information and, by doing so, "communicate" in a rudimentary way. The decisive factor in the characteristic of world-openness is that a natural being, due to its organic endowment (brain, nerve cells), perceives something of its environment and can respond to environmental stimuli. The more complex an organism is structured the more agency aspects are revealed in such response. Expressive behavior and intraspecies communication count as strong evidence for "world-openness." These criteria, however, do not entail ecosystems as such in the moral community.

All physiocentric positions (as sentientism, biocentrism, ecocentrism, and holism) can be either gradual or egalitarian. With high likeliness, the egalitarian-gradualism-divide is as crucial for the demarcation problem as the criteria themselves. Egalitarianism claims that all members of the moral community have the same inherent value. The rationale, however, is less clear than the claim. Neither does the moral point of view imply egalitarianism, nor is egalitarianism a conceptual truth of inherent moral value.

Gradualism claims that morally relevant traits of natural beings (sentience, consciousness, world-openness) come and go by degrees. This gradual scale of morally relevant properties itself permits grading. Even the species-specific proliferation strategies (K- versus r-strategies) can make a difference with respect to single tokens (mice, frogs, fish). Grading is close to the ways organisms live. If less than 1% of the newborn tokens of specific species reach the adult form, a single life does not count that much. Respect for nature also means to respect evolutionary traits. This holds true for marine life. There should be leeway for grading between marine mammals, marine birds, turtles, sharks, sardines, crabs and shrimps, shellfish, molluscs, plankton, etc.

A close examination of the demarcation problem enables moral agents to distinguish between natural entities which are appreciated for the services they bring about and other entities which must be respected morally for their own sake. As Muraca (2011) shows, there are many options to combine appreciation and respect. As Norton (1991) argues such combinations of appreciation, intertemporal fairness, and moral consideration constitute practical-political convergences in nature conservation policies despite remaining ethical disagreement.

19.3.7 Conclusion

ESA is a highly useful tool for bridging the gap between ecosystems and human values. It allows for measurement, quantification, and economic evaluation of crucial ecosystem services. ESA studies bring about robust results in terms of physical or monetary units. ESA is, however, limited in scope and method. It faces methodological limits with respect to obligations and commitments, intergenerational equity, spiritual services, biophilia, virtues, and inherent moral values. If, however, ESA is connected to these six lines of environmental ethical reasoning, it can and should become an eye-opening device for the deeper layers of environmental ethics. Therefore, ESA works as a turning-table. On the one hand, it makes the contributions of ecosystems to human welfare visible and can calculate such welfare effects in economic terms. On the other hand, it can serve as an entrance gate for environmental ethics. ESA can and should be used as such turning-table between scientific support for environmental policy-making and ethical reflection.

If such turning-table function is recognized, it can help to address problems of conflicts and trade-offs. ESA, without amendments, is silent about how conflicts and trade-offs should be decided in case of conflict. A theory of environmental conflicts is suggested but not entailed in the ESA. Should humans produce more provisioning services at the expense of cultural ones or should they better reverse the trend to produce provisioning services at the expense of cultural ones? ESA seems to be neutral in this respect as it leaves the trade-offs between kinds of services to the market, to stakeholder negotiations, and to political decision-making (see Bromley and Paavola 2002). Environmental ethics might encourage ESA-scholars to defend underrated cultural services against the widespread dominance of provisioning and regulating services. A concept of conflict resolution is beyond the scope of this subchapter. It must suffice to say that ESA should take a view in the world of normative orders, as entitlements, rules, obligations, and commitments.

19.4 Systems-Based Aspects of Human–Environmental Relations

All the human factors discussed above are also strongly influencing the environment, generating solid interrelations between the human and non-human system elements. The resulting entities will be briefly and generally characterized on the following pages, guided by the question for the relations, the flows between the pools and their consequences from a system-analytical viewpoint. These constellations are applied to coastal conditions, and in the end, some concrete human elements of the Baltic human-environmental systems will be briefly identified.

Discussing these items, we are moving forward from the ecosystem conditions described in Chaps. 2, 3, and 4 into coupled human-environment systems (HES or CHANS as coupled human and natural systems, Chen 2015), which characterize the dynamical interactions between human systems and natural entities (Sheppard and McMaster 2004; Liu et al. 2007; Alberti et al. 2011; Scholz and Binder 2011). This linkage expresses the idea that the mutual evolution of humans on the one hand and environmental systems on the other—especially in the anthropocenic age or under the target of sustainable development—should not be treated as individual, isolated systems. Instead, the concept of human-environmental systems (also termed social-ecological systems, coupled human and natural systems, or coupled human-bio-physical systems; see e.g., Chapin et al. 2009, Chen 2015) recognizes that the social, economic, and cultural well-being of people depends not only on their relations with other people, but with the physical and biological environment as well. These relations often describe the environment as stocks of resources as well as the capacity of the environment to function as a life support system, providing several ecosystem services.

Following Colding and Barthel (2019) human-environmental systems (HES) are complex adaptive systems (Müller and Li 2004). They provide key characteristics such as: (1) integrated biogeophysical and socio-cultural processes, (2) self-organization, (3) nonlinear and unpredictable dynamics, (4) feedback between social and ecological processes, (5) changing behavior in space (spatial thresholds) and time (time thresholds), (6) legacy behavioral effects with outcomes at very different time scales, (7) hierarchical structures and emergent properties, and (8) the impossibility to easily extrapolate the information from one SES to another" (Colding and Barthel 2019).

The basic quantifiable features of these systems are the flows of energy, water, matter, and information. These subjects can be organized in different quantities, qualities or utilities, the flows can be triggering growth and development as well as disturbance and decay, they can accelerate service provision or disservice impacts, they may be supporting the systems' integrity or provoke degradations of the human-environmental entities.

A generalized depiction of these HES can be seen in Fig. 19.5. On the one hand, the sketch demonstrates the distinction of human and environmental subsystems referring to some of their basic elements. It is easily visible that the internal components, their structures, linkages, and subsystems are extremely different.

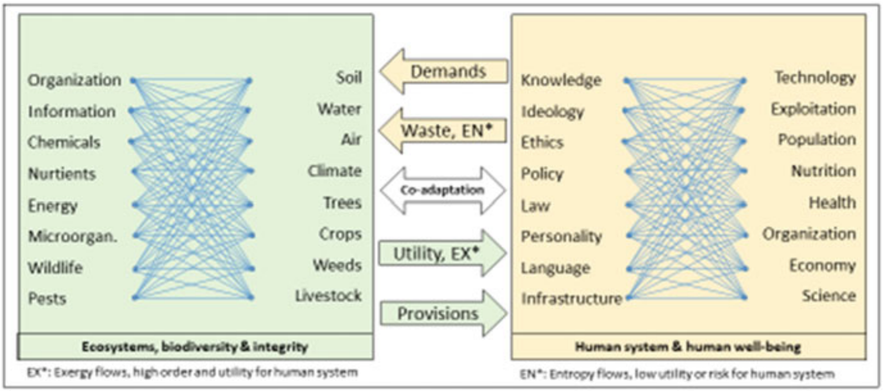

Fig. 19.5 Basic elements of human-environmental systems and fundamental characteristics of interacting flows after Marten (2001)

And on the other hand, the basic interrelations between human and natural subsystems, which can comprise an enormous complexity, are characterized here in the following scheme: Human subsystems are developing and expressing demands for ecosystem services from the natural units (see Chap. 2), including all of their classes in local intensities and sequences. Contrary, the ecological entities are able to provide the respective ESS. Most of the resulting nature-culture-flows include transfers of relatively ordered structures, which provide a relatively high capability to be transformed into mechanical work or utility, thus owing a high degree of exergy (Joergensen and Müller 2000; Nielsen et al. 2019). The opposite direction (flows culture-nature) often is accompanied by high degrees of entropy, disorder and waste which flow back to the nature side after a degradation within the human-technological networks.

In order to better relate these multiple components within a generalized sequence of causes and effects, the DPSIR approach has often been successfully used as an approved instrument of integration between human and environmental processes and structures (Smeets and Weterings 1999; Borja et al. 2006; Svarstad et al. 2008; Burkhard and Müller 2008; Gari et al. 2015). The idea is that the society implies social, demographic, and economic developments and corresponding changes in lifestyle which influence the levels of consumption and production and which strongly influence the motivations of the acting persons for specific land use strategies. These drivers (D) are responsible for the production of pressures (P), the release of substances, physical, chemical, and biological agents into the ecosystem by the resource and land use realization. As a consequence the condition or the state of the ecosystem (S, measurable by biological and ecological indicators) can be modified, and this will have potentially disturbing impacts on the ecological (I1) and human (I2) subsystems. After a reception of these disarrangements, the society can carry out actions to minimize the negative impacts imposed on the environment (response R). In Fig. 19.6 these causal hypotheses are arranged in relation to the ESS

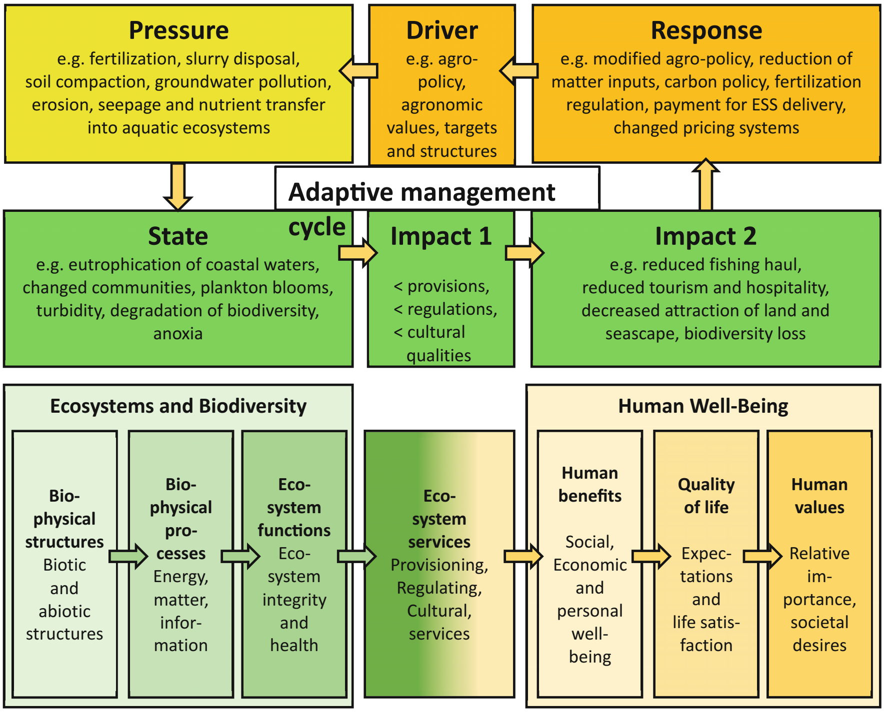

Fig. 19.6 Linking the ecosystem service cascade (see Fig. 19.1) and the DPSIR indicator approach for human-environmental systems

cascade which has been introduced in Chap. 2 of this volume. In Fig. 19.6, eutrophication has been chosen as a case study to demonstrate these human-environmental interrelations. Here the abiotic elements and the biodiversity components are producing ecosystem functions. All of these environmental activities can be observed as parts of the state function S. This situation is based on certain societal drivers (D, e.g., agro-policy), which provoke pressures that are responsible for the state dynamics (P, e.g., fertilization). The ESS flow can be understood as a first impact on the ecological element, if a modification of the functionality entails a decrease of ESS capacities. Such a new development will be recognized by the society as it leads to a decrease of human well-being (Impact 2). And consequently, there should be a political or administrative or management-based reaction, e.g., a change of the fertilization policy (R).

What we can keep in mind from the sequence of figures is that there is an extraordinary high complexity of relationships between coastal, human and ecological subsystems. While searching for sustainable pathways for future development, it is obvious that the interactions provide a certain directionality with exergetic flows to society and entropic flows transferred to the ecosystems. In order to understand these unilateral linkages, the DPSIR approach of the European Environmental Agency can be adopted to underline the interactions within the causal chain of an adapted

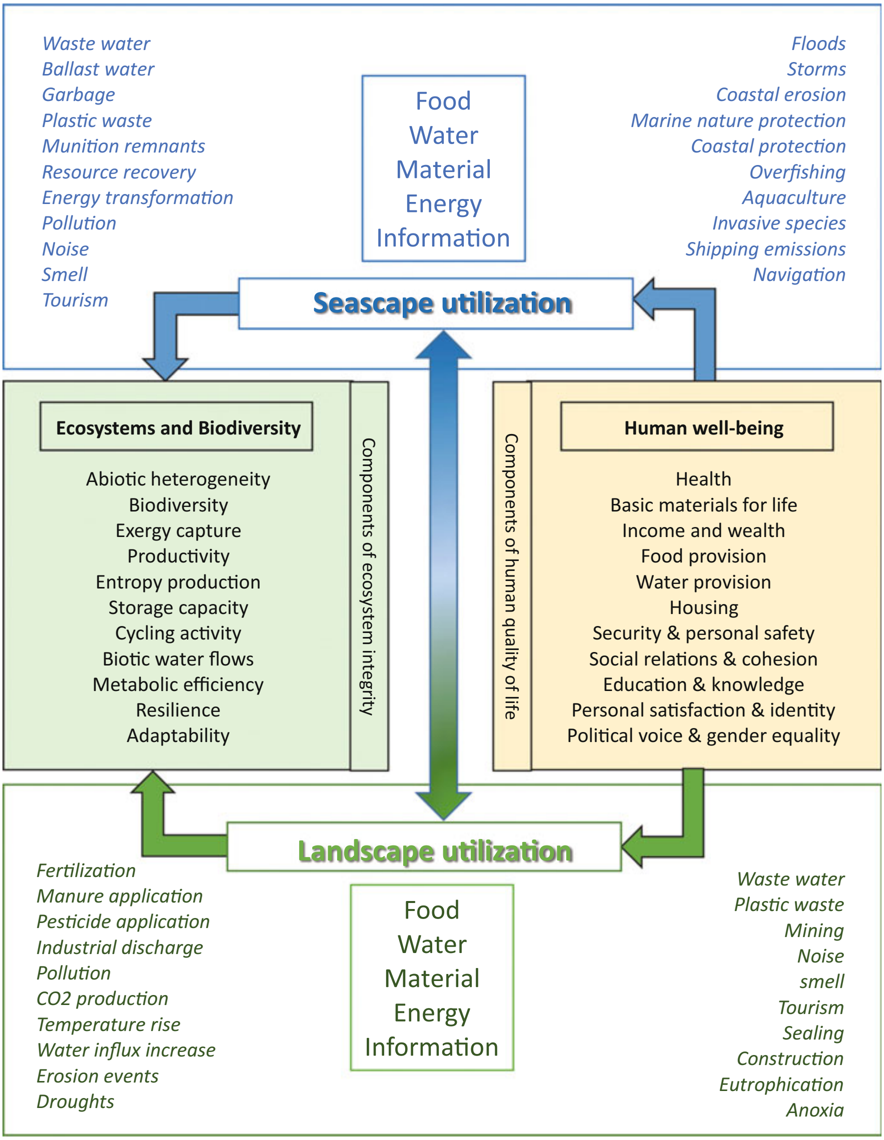

Fig. 19.7 Flows and pressures (*italic*) from land and sea toward coastal ecosystems, some of their basic parameters of systems conditions, and some resulting effects

management model. If we convey these conditions to the marine-terrestrial environments of this volume, things might move to another step of complexity, because in general two different directions of the pressure-based networks are possible, one resulting from direct seascape utilization and the other originating in landscape resource use. Figure 19.7 shows some of the many related pressures, the focal integrity parameters of the ecosystem and some features of human well-being, which are the target values of the ecosystem service provision. We have to be aware that all of the involved interactions are active, resulting in an enormous functional network. But there might be some more significant and some less important influences in a certain study site. Therefore, it makes sense to take a look at the effectual boundary conditions of the concrete research area before the exchanges of ecosystem services are discussed.

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Introduction: The Concept of Ecosystem Service Assessment Applied to Coastal Systems

20

Felix Müller and Hendrik Schubert

Abstract

This short chapter includes an introduction into the concept of ecosystem services. Thereby, we bring together the ecosystem service definitions, which have been mentioned in the chapters ahead, and we discuss ecosystem service types and classifications as well as the basic ideas on ecosystem service production. Finally, we provide an outline about the respective articles, following on this subject.

This whole volume might be understood as a bundled publication targeted on an improved comprehension of the ecosystem service approach: While Part 1 has prepared the mono, multi-, inter- and transdisciplinary grounds of the involved scientific branches, the regional boundary conditions for the spatial ecosystem service settings are narrated in Chaps. 3 to 8. The eco-physiological ecosystem conditions, important flows, stores and diversities of ecological structures and functions are described in Chaps. 9 to 18, and some significant features of human influences, philosophies and comprehensions of ecosystem services as well as ethical attitudes are recounted in Chap. 19. Therefore, several terms and relations concerning ecosystem services, have been explained and depicted above, e.g. the hierarchy of needs, which demonstrates the target functions of the service approach (Fig. 19.1), or the cascade model integrating ecosystem condition, ecosystem service and human valuation (Fig. 19.1). In addition, basic definitions (Table 19.1) and

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classification schemes were discussed above (e.g. Chap. 19.2), and the methodological demand for a matrix approach has been pronounced in Figs. 19.2 and 19.3. Consequently, the following introduction can be short and it can be reduced to a brief statement of focal specialities of ecosystem comprehension within the group of contributors of this book.

20.1 Comprehending Ecosystem Services

In Table 19.1 it was shown that—as expected—several different approaches are existing to define ecosystem services. As a summary of that description, ecosystem services can be understood as sets of environmental properties deriving from ecosystem structures and processes, which are arranged from an anthropocentric point of view: They describe those products and outcomes from complex ecological interrelations, which are useful and necessary for human well-being. Ecosystem services are the benefits people obtain from ecosystems, and thus they are also used to represent the environmental interrelations between the three sectors of sustainability. The "working definition" of this chapter has been adopted from the simple and consensual statement of the 2010 Salzau ESS conference: "Ecosystem services are the contributions of ecosystem structure and function—in combination with other inputs—to human well-being" (Burkhard et al. 2012).

Due to the title of this section, we also want to shortly mention the term "assessment". Following Potschin-Young et al. (2018), it is "the analysis and review of information derived from research for the purpose of helping someone in a position of responsibility to evaluate possible actions or think about a problem". Assessments include assembling, summarizing, organizing, interpreting, and possibly reconciling pieces of knowledge and communicating them. An ecosystem assessment is understood as "a social process through which the findings of science concerning the causes of ecosystem change, their consequences for human 'well-being', and management and policy options are brought to bear on the needs of decision-makers". In so far, several of the following valuation studies are methodological preparations of approaches and tools with reality tests, in order to be adopted and applied in decision-making processes.

These contributions of nature for human well-being can be classified from many viewpoints. In the following chapters, the service types from Kandziora et al. (2012, see Table 20.1), Burkhard et al. (2014) and the CICES distinction (*Common International Classification of Ecosystem Services*, see Haines-Young and Potschin 2018) have been mainly utilized. As in most recent classifications, three groups of services are distinguished: regulating, provisioning and cultural services. Regulating services are the benefits people obtain due to the regulation of natural processes and the control or modification of biotic and abiotic factors. Provisioning services comprise all material outputs from ecosystem processes that are used use for human nutrition, processing and energy use. These products can be traded and consumed or used directly, thus they are the desired 'end-products' of nature providing clearly visible benefits to society. Cultural ecosystem services are the

Table 20.1 List of important ecosystem services used in the following papers of this volume

Provisioning services ... all material outputs from ecosystem processes that are used for human nutrition, processing or energy	Crops (human nutrition)
	Biomass for energy
	Crops (fodder)
	Livestock
	Timber
	Fibres
	Wood fuel
	Wild food
	Fish and seafood
	Flotsam and algae
	Ornamentals
	Drinking water
	Abiotic energy
	Minerals
Regulating services ... the benefits people obtain due to the regulation of natural processes and the control or modification of biotic and abiotic factors	Groundwater recharge, water flow
	Local climate regulation
	Global climate regulation
	Flood protection
	Air quality regulation
	Erosion regulation, wind
	Erosion regulation, water
	Nutrient regulation
	Water purification
	Pest and disease control
	Pollination
Cultural services ... the intangible benefits people obtain from ecosystems in form of non-material, spiritual, religious, inspirational and educational experiences	Recreation and tourism
	Landscape aesthetics + inspiration
	Knowledge systems
	Cultural heritage
	Regional identity
	Natural heritage

intangible benefits people obtain from ecosystems in form of non-material spiritual, religious, inspirational and educational experience. These services provide benefits for human recreation and mental and physical health, experience by tourism, aesthetic appreciation and inspiration for culture, art and design, spiritual experience and sense of place.

20.2 Conceptualizing Ecosystem Service Production

While the functional quality of an ecosystem can be described unvalued by integrity variables or state indicators, ecosystem services have to provide a contribution to human well-being; there must be a demand for the results of the respective environmental processes. Thus, ecosystem services are focal components of the transfers within human-environmental systems. In order to better comprehend these complex relations, many authors have constructed conceptual frameworks for ecosystem service assessments. Figure 20.1 shows the so-called ecosystem service cascade after Haines-Young and Potschin (2018), which is the most frequently used framework today. It demonstrates a functional hierarchy of ecosystem processes and structures, which is ordered to focus on the contributions of ecosystem relations for providing human benefits: All the multiple objects of ecological investigations may refer to the structures as well as the processes in an ecosystem. These items are bundled in the set of ecosystem functions, which are able to derive the potentials of an ecosystem to provide a certain service as a result of intensive interactions between structural units and processes. The functionality of an ecosystem can be indicated by its health or integrity or sets of other developing state variables (Nielsen et al. 2019). The functions are turned into services if they are utilized to produce a benefit related to social, economic or personal well-being factors. Consequently, services are groups of functions that are selected due to their utility for human society (Müller et al. 2015, Schneiders and Müller 2017).

If these services have a high significance, they will receive a high societal value, and their relative importance will be highly considered in human-environmental trade-offs. These values will be different at different places as the demands for the

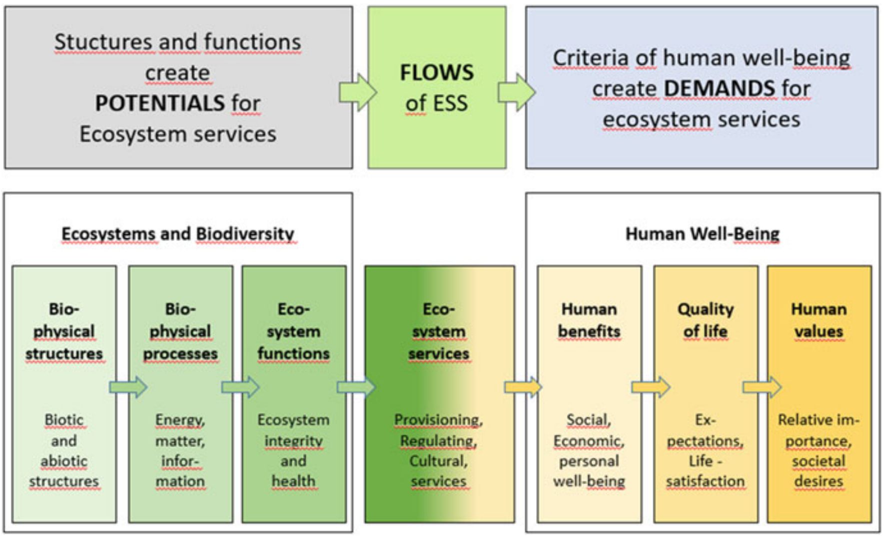

Fig. 20.1 Distinction of ecosystem service features

mentioned benefits are varying spatially due to the special site conditions. They will be of different significance for different groups of people due to their specific objectives and backgrounds. They will furthermore be different due to varying degrees of ecological comprehension, and there will be temporal differences due to the dynamics of special pressures on sustainable developmental pathways (see also Chap. 19).

20.3 Introducing the Contents of Chap. 21 to 26

After this short introduction to the topic of ecosystem services, T. Kuhn and colleagues provide a review article on the scientific ecosystem service situation around the Baltic Sea, demonstrating missing links and addressing recent gaps of knowledge (Chap. 21). Some of these gaps are filled in the following articles, which illuminate spots of ecosystem service research in the projects BACOSA and SECOS. K. Ott and M. Berg show some qualitative results of their works on cultural coastal ecosystem services in Chap. 22, and K. Frank and M. Benkenstein present an economic exercise on the monetary valuation of coastal land- and seascapes from the viewpoint of tourists and coastal inhabitants (Chap. 23). In Chap. 24 J. Schumacher and colleagues demonstrate an ecosystem service matrix approach which is used to create maps of spatial service distributions around the coastline. Furthermore, this approach is applied in Chap. 28 in order to evaluate the influences of phytoplankton, macrophytes and bioturbation on ecosystem service production. The following paragraphs are related to temporal developments of ecosystem services: While Chap. 25 by M. Inacio and G. Schernewski concentrates on historical dynamics, Chap. 26 is focussing on potential future traits of ecosystem services potentials.

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The Missing Links in Ecosystem Service Research

21

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Abstract

The marine and coastal ecosystems of the Baltic Sea are exposed to an intensification and diversification of anthropogenic activities and related environmental pressures. Human interest in marine resources and space often overlap with environmental protection objectives, causing conflicts. Research can assist capacity building to enable knowledge-based decision-making in marine management and policy to help solve these issues. Three participatory systematic maps were carried out on marine and coastal ecosystem services (ES), monetary and non-monetary valuation methods applied to value them, and the interrelation of ES and human health and well-being in the Baltic Sea region. Policy advisors were engaged throughout the review process. The aim was to map existing

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scientific knowledge and identify knowledge gaps for the scientific community and to support the implementation and update of the key marine protection policies in the region. This chapter introduces the review methodology, provides an overview of knowledge gaps and missing links in ES research, and addresses future steps to connect the dots.

21.1 Marine Policies and the Ecosystem Approach

Marine policy and management decisions in the Baltic Sea predominantly target the condition of ecosystems in order to regulate anthropogenic pressures and meet the environmental objectives. The commonly used framework is the ecosystem approach adopted by the Convention on Biological Diversity (CBD) to support "integrated management of land, water and living resources", thus emphasizing the "intrinsic value" of biodiversity and ecosystems (UN CBD 1992).

Since 1974, the Baltic Sea Marine Environment Protection Commission (HELCOM) has coordinated the environmental protection objectives as well as environmental assessment and management goals of the nine littoral Baltic Sea countries and the EU. In 2007, HELCOM adopted the Baltic Sea Action Plan (BSAP) with the aim to reach good environmental status in the Baltic Sea by 2021. This plan included objectives on eutrophication, biodiversity, hazardous substances and maritime activities. To reach these objectives, HELCOM established the Group for the Implementation of the Ecosystem Approach (GEAR) dedicated to marine management and the sustainable use of marine environments. In addition, an expert network of economic and social analysts specializing in the use of marine waters and the cost of degradation was formed to provide recommendations and advice to the HELCOM GEAR group (HELCOM 2018a). The European Union Marine Strategy Framework Directive (EU MSFD) (Directive 2008/56/EU), however, is the key policy on environmental protection of marine ecosystems and was established in 2008. It extended the EU Water Framework Directive (WFD) to all European waters to achieve Good Environmental Status (GES) by 2020.

The BSAP and the European Union Maritime Policies, including the MSFD and the Maritime Spatial Planning Directive (MSPD) (Directive 2014/89/EU), apply the ecosystem approach for the integrated management of marine resources. Through the MSFD, 11 qualitative descriptors were established that describe the ecosystem condition when the GES has been achieved. Similarly, the goals and objectives of the BSAP represent its main aims by linking the environmental status and the impact of anthropogenic pressures to the environment. Due to the main target of policy objectives and management decisions to achieve and maintain GES of marine ecosystems, the research focus has been predominantly on ecosystem processes and functions (e.g. Lindh and Pinhassi 2018; Carstensen et al. 2020) and the development of assessment methods and environmental indicators to assess the environmental status of ecosystems (e.g. Borja et al. 2013, 2014; Lyons et al. 2017). The MSPD, and likewise the MSFD, require the maritime spatial plans and the Marine Strategies of all EU member states to consider ES to achieve and maintain healthy marine ecosystems and to enable their sustainable use (Article 4 MSPD; Article 1, MSFD). While the official reporting has a mandatory requirement for the assessment of the environmental status, it makes the assessment of ES and the application of the ES approach obligatory only for the economic and social analysis. Due to these limited institutional requests, the application of the ES concept is not well-developed in marine and coastal ecosystem management and decision-making (Boulton et al. 2016; Drakou et al. 2017).

However, considering the manifold interrelations of human actions and the condition of nature, sustainable management cannot simply focus on the ecological status, but needs to analyze and integrate all aspects of the socio-ecological systems. Therefore, three participatory systematic literature syntheses on (1) marine and coastal ecosystem services research in the Baltic Sea, (2) monetary and non-monetary valuation methods used in the region, and (3) the relationship of the Baltic Sea ecosystem services and human health and well-being were carried out to provide an overview of the available scientific knowledge on marine and coastal ES research in the Baltic Sea region. In this way knowledge gaps were identified, and the available scientific evidence made more accessible for policy makers and the scientific community alike. This chapter constitutes a summary of the outcomes of the three studies (Håkansson et al. 2020; Kuhn et al. 2021; Storie et al. 2021).

21.2 Participatory Systematic Mapping of the Evidence Base

Systematic literature mapping is a method to review literature with the aim to identify, collate and describe the evidence base to a specific question and identify research gaps in a repeatable and objective manner. First applied in medical research, the Collaboration for Environmental Evidence (CEE) developed guidelines (CEE 2018; Haddaway et al. 2018) that set standards for synthesizing environmental scientific information for decision making. Literature is reviewed under transparent conditions and with pre-defined criteria to reduce bias. Stakeholder involvement is considered beneficial for the process of planning and conducting systematic maps, and to support the development of policy relevant outputs that enable decision-making with the best available knowledge (Haddaway et al. 2016, 2017).

Figure 21.1 gives an overview of the review questions and illustrates the screening process of the three participatory systematic maps. The search string development included test searches to validate the comprehensiveness of the search strings, composed of geographical keywords, ecosystem/ ES keywords and synthesis-specific keywords, against benchmark lists of publications that were previously defined as relevant through expert knowledge and snowballing. Searches were carried out in multiple bibliographic databases and search engines (e.g. Web of Science Core Collection, Scopus, BASE) and after duplicate removal, publications were subsequently screened for relevance on title, abstract and then at full-text level based on pre-established inclusion and exclusion criteria. All levels of screening and data extraction were carried out by teams of two or more reviewers and consistent

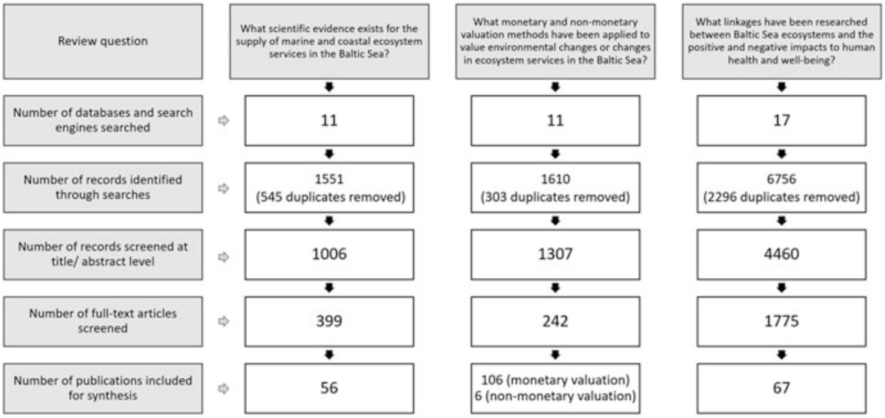

Fig. 21.1 Review questions and number of publications throughout the screening process

practice was ensured by double screening and coding a subset or respectively all publications to consolidate repeatability. After the collection and collation of publications, data was extracted and synthesized. Policy advisers from the HELCOM GEAR group were involved throughout the review process, e.g. to define the scope of the research, discuss the interim synthesis results and to clarify the policy relevant main messages from all syntheses (Kuhn et al. 2021). For more detail on the methodological approach and overviews of all relevant publications of the three syntheses, please see (Storie et al. 2020, 2021; Håkansson et al. 2020; Kuhn et al. 2021).

21.3 Ecosystem Service Research

Following the general trend in the past decade, research on marine and coastal ES has been a growing field in the Baltic Sea region. Studies have been mainly focussed on regulating ES (37.2%) with a special focus on the regulation of nutrients. Research on cultural ES (32.6%) and provisioning ES (30.3%) were predominantly represented by studies on the provision of fish and recreational aspects of the human interaction with nature. More than 80% of studies only considered ES supply and did not incorporate ES demand. 40.7% of publications focused on biophysical studies in comparison to social (18.5%), economic (18.5%) and management/ policy approaches (22.2%). Typical study designs were economic (17.8%) and biophysical assessments (16.8%), modelling approaches (16.8%) as well as surveys (15.8%) and expert assessments (10.9%) (multiple answers possible). Studies on the development and application of ES indicators and therefore publications that systematically apply ES assessment and mapping approaches, as requested by MAES (Mapping and Assessment of Ecosystems and their Services), Target 2 Action 5 of the European Biodiversity Strategy to 2020, are rare (5 studies) (e.g. Veidemane et al. 2017; Ruskule et al. 2018; Depellegrin et al. 2020). The understanding of how changes in ecosystem properties and functions cumulatively affect the ability to supply ES is limited as studies mainly focus on specific aspects of the ES cascade model (Potschin and Haines-Young 2011). Neither do these studies necessarily link human actions, biophysical structures and processes via ecosystem functions with ES and the impact on the benefits humans gain for their health and well-being. This knowledge is crucial to assess the vulnerability of ecosystems to the numerous human activities associated with the Baltic Sea. Conversely the integration of the influence of drivers of change, like anthropogenic pressures as well as policy actions, on ES supply would be valuable. While there are a few publications that consider the supply of more than one ES (e.g. Troell et al. 2005; Ahtiainen et al. 2019; Viirret et al. 2019), studies on ES trade-offs and synergies or on the interactions within ES bundles are missing. ES research in the Baltic Sea is characterized by limited use of a classification system. 24.5% of publications applied the Common International Classification of Ecosystem Services (CICES) (Haines-Young and Potschin 2018), while another 7% of studies referred to the four ES categories established by the Millennium Ecosystem Assessment (MA 2005). The restricted application of systematic classifications in the scientific literature makes room for an inconsistent use of terminology and leaves space for misinterpretation.

The word clouds in Fig. 21.2 display synthesized the categorized terms of all provisioning, regulating and cultural ES mentioned as examples of ES in the publications. Word size resembles how often the respective category is used, e.g. larger words were mentioned more often or represent a category of word clusters. The main word clouds in the centre depict the categorized major findings, while the smaller word clouds on the right display a more detailed account of chosen categories. Comparison of word size and therefore the frequency of appearance is only valid within each cloud. The word clouds show, on the one hand, the plethora of terms for one ES, e.g. related to nutrient mitigation. On the other hand, they indicate the broad notion in which some ES, e.g. the provision of fish, are discussed. For example in the use of the term “fishery” as an ES, there is no consistent differentiation between the service supplied by the ecosystem and the human action to extract the resource. As for the application of CICES, the word clouds indicate that the classification is most often applied to identify regulating ES. In addition, space and biodiversity, which are not considered as ES in CICES, are often discussed as such. This analysis and the stakeholder involvement carried out during the review process indicated, that more emphasis should be given towards developing a more consistent terminology within the research community, as well as reconsidering how the communication towards stakeholders and the general public can be simplified.

21.4 Ecosystem Service Valuation

A systematic mapping of scientific literature of monetary and non-monetary valuation methods was set up to get an overview of which valuation methods have been used to value the benefits of an environmental/ES improvement in the Baltic Sea or the costs of not reaching the environmental protection goals of the Baltic Sea

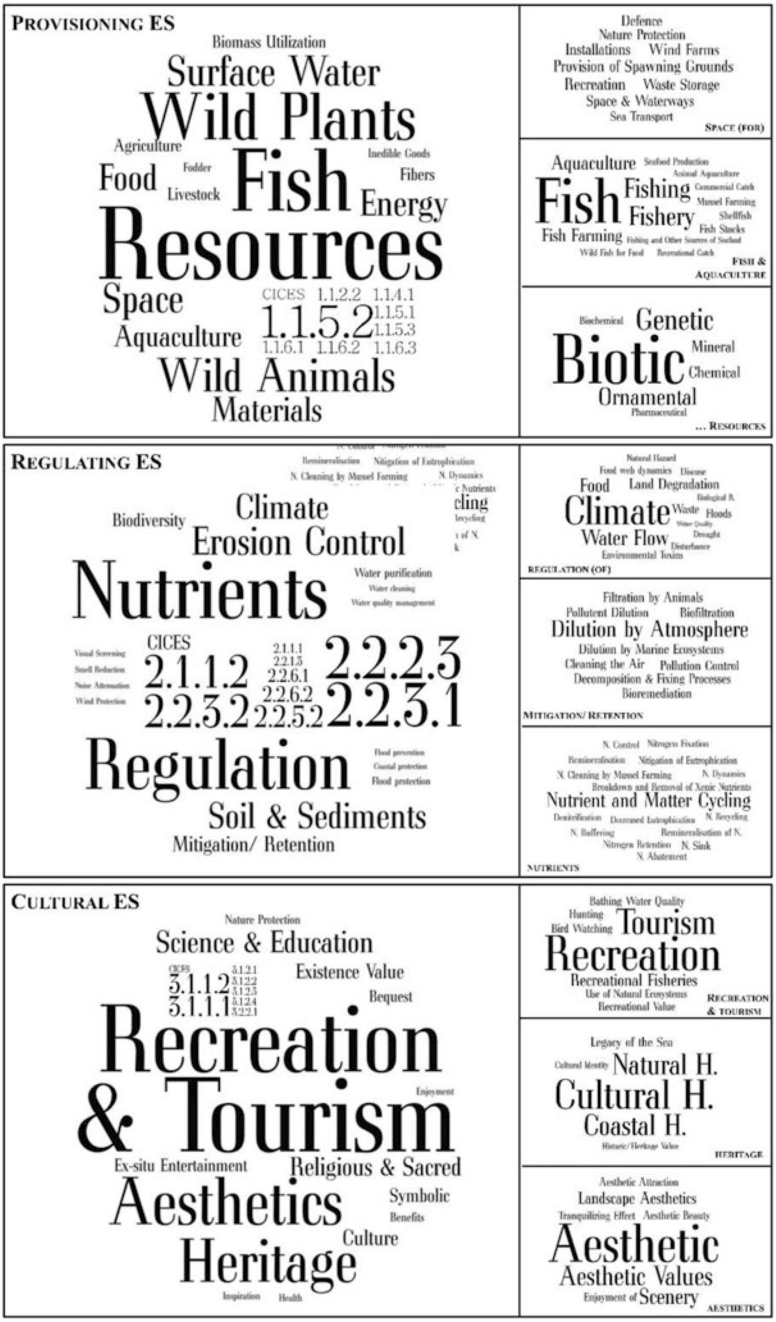

Fig. 21.2 Word clouds displaying the categorized findings of provisioning, regulating and cultural ecosystem services mentioned

(Håkansson et al. 2020). Monetary and non-monetary valuation methods capture people's preferences, perception and motivations and measure these using quantitative, semi-quantitative or qualitative value indicators (e.g. Ninan 2014). The choice of the right valuation method to be used depends entirely on the need for information, i.e. the question to be answered. So-called cost-based methods can be used if the aim is to find the cost of reaching/not reaching an environmental/ES improvement. However, if policy makers are particularly interested in getting to know how the citizens value the benefits of environmental/ES improvements, or avoidance of degradation, then methods that can capture people's preferences need to be applied (benefit-based methods). The most straightforward way of approximating how people's well-being is affected by a policy action is to use a method that is based on market prices (marked-based methods). (e.g. Hanley and Barbier 2009) The results from non-monetary valuation methods can be used for various purposes, without using monetary metrics, from solving conflicts between different stakeholders to assessing the acceptability of environmental programmes. (e.g. Santos-Martín et al. 2018).

Notably our results showed that, although a number of different valuation methods were available, the cost-based method abatement cost (44%), and the benefit-based method choice experiment (40%) dominated the applied monetary valuation methods to a large extent. Conspicuously only six applications of non-monetary valuation methods were identified. Two major gaps identified in our mapping were that valuation research did not apply the ES concept and they did not make the connection to marine protection or other marine policies. For example only 13% of the studies applying monetary valuation methods used ES as a keyword in their research article. Although the authors of the research articles did not seem to apply the ES concept in their study, the researchers conducting the systematic mapping were able to apply CICES to the existing studies and interpret which ES were valued. Nearly 60% of the monetary valuation applications studied cultural ES, whereas only 11% of the studies considered regulating ES. The MSFD, that is the policy explicitly calling for economic analysis and the use of ecosystem approach, was mentioned only in 16% of the studies.

As pointed out in Sect. 21.3 there is a lack of ES research, and evidently, if the impact of a marine policy to the marine environment is not known in biophysical terms, the basis for valuation is not solid. Increasing the number of marine ES assessments where the impact of a policy to the magnitude of the ES supply is studied would facilitate the valuation of the ES. Also, valuation studies considering single ES are probably less valuable for decision-making since they do not provide the basis for analyzing trade-offs between different ES. Further, it is important to point out that in order for the valuation results to be used in policy making it must be clear to the policy makers what has been valued (both in terms of ES/environmental change and in terms of what a valuation method can and cannot capture). Hence, we argue that an effective marine ES valuation requires interdisciplinary collaboration and science-policy dialogue.

21.5 Human Health and Well-being

Health and well-being have a range of definitions in the literature from the functional use of proxies, such as life expectancy, to a more holistic understanding that utilizes a range of factors (Storie et al. 2020). However, the knowledge of the ES that the Baltic Sea provides to the health and well-being of those who live in the region or visit it, is lacking. While human populations have had a significant negative impact on the Baltic Sea ecosystem, which are well documented (HELCOM 2018b), the positive and negative impacts of the Baltic Sea on human populations are not as well elaborated in the scientific literature, as the linkages are often vague and lack detail (Storie et al. 2021). For example articles may mention human health is negatively impacted by the ecosystem, but do not elaborate on the specific health impacts.

Society protects what it values but does not protect what it does not understand, therefore there is a need to understand the benefits that the Baltic Sea provides to human populations and the consequences of environmental degradation on human health and well-being. Literature suggests that improving the knowledge within society also improves the acceptability of the measures taken to restore and protect ecosystems (Pakalniete et al. 2017; Schernewski et al. 2018; Thomas et al. 2018; Hyytiäinen et al. 2019). Once people understand that a good environmental status is good for their health and well-being, they are often more supportive of the measures taken.

The systematic literature search showed there are articles focused on health issues arising from exposure to the Baltic Sea, such as cancers from eating fatty fish (Hagmar et al. 1992; Glynn et al. 2013) or infection from antibiotic-resistant organisms (Literak et al. 2010; Mudryk et al. 2010; Bier et al. 2015). Studies were also carried out that documented how degraded ecosystems are leading to poor health and well-being outcomes for society (Ahtiainen and Öhman 2014; Veidemane et al. 2017; Nieminen et al. 2019).

Few articles, however, bring these aspects together in any detail. Those that explicitly mention ES rarely provide examples of the health and well-being impacts of ES on people, they merely mention the potential for impacts. Those articles that do mention ES tend to focus on the benefits of cultural ES such as recreation (Czajkowski et al. 2015; Ahtiainen et al. 2019; Bertram et al. 2020) or the provision of fish for good nutrition (Veidemane et al. 2017). In addition, knowledge of the impacts of the Baltic Sea on health and well-being is scattered across multiple disciplines. For example detailed effects on health arising from exposure to the Baltic Sea ES are found primarily in the medical literature, however, these papers do not link to the ES concept.

There is a need for a common understanding of the benefits, not just health, but the full range of well-being benefits that the Baltic Sea ES provide. The benefits include economic and material contributions to living standards; healthy food; security and safety of users through coastal protection; social relations, governance and freedom of choice and action connected to how the Baltic Sea's resources are used and enjoyed; subjective well-being and culture related to the aesthetic and recreational opportunities the Baltic Sea provides and so on. There is a need to understand these impacts on human health and well-being because they are not always obvious to society and therefore education is needed. For example, knowledge is limited on the multiple benefits provided by coastal wetlands, which include the provision of clean water, maintaining healthy beaches and reducing erosion. Often society thinks reeds by the beach are unaesthetic and do not understand the benefits. Society needs to see the beauty in the complexity of the ES provided by the Baltic Sea and thus be able to value the benefits.

21.6 Implications for Research to Support Environmental Management and Policy

The main marine policy focus and thereby the leading research emphasis has been on the GES of the Baltic Sea ecosystems and therefore significant knowledge gaps on ES exist. Our society is driven by limitless economic growth, while ecological resources are limited. Therefore, the management intention based on the ecosystem approach could obtain depth by involving the complex linkages and dynamic relationships between human pressures, biodiversity, ecosystem condition and the supply of ES. As environmental protection constitutes only one aspect of social decision-making and is not the highest priority, environmental management of ecosystems needs to balance the status of ecosystems with anthropogenic interests and importantly, underline the importance of healthy ecosystems for human existence. To support evidence-based decision-making that integrates environmental protection and human use of the marine environment, increased efforts are needed to assess and quantify ES and their synergies and trade-offs, that builds the foundation for ES valuation. The lack of a standardized terminology and classification within the research community was identified as source for misunderstanding, as well as an obstacle to develop a common approach for the communication towards policy makers. In addition, there is a need to communicate more effectively to the public to help them understand the value that ES provide for them. The involvement of a broader range of stakeholders (e.g. concerned citizens, ES users, funding agencies and policy makers) and a strong focus on transdisciplinary ES research that incorporates ecological assessments, environmental management, as well as medical and socio-economic research is needed to support sustainable development for the Baltic Sea.

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22 Eudaimonic Valuation of Cultural Ecosystem Services

Konrad Ott and Margarita Berg

Abstract

Chapter 5 argued that cultural ecosystem services should not be underrated in ESS-assessment. This sub-chapter substantiates this claim by a specific case study as it deals with those cultural services as they originated and established in specific historical periods at the Baltic Coastline. The focus of the research project this sub-chapter is based on recreation, aesthetics, natural heritage, and knowledge systems. The specific features of cultural services (perception, symbols, etc.) are reflected upon. The method and the scope of sources are outlined. There are several findings which highlight the importance of specific cultural services for the Baltic coastlines in past and present times. Findings refer to the different cultural services within tourism, the values of atmospheres, moods, and sceneries in landscape painting, and some recent ideas to provide new ways of access to nature by specific trails. These values are framed with respect to different periods in the history of German nature conservation but also in Germany's general history. The conclusion points to crucial tasks for coastal management.

22.1 Introduction

In addition to provisioning and regulating services, the so-called cultural ecosystem services can also be studied. The theoretical foundations given in Chap. 5.3. from an environmental ethical perspective pointing at the parallels between cultural ecosystem services and eudaimonic values. Since the scope of ecosystem cultural services is broad and the research project took different historical periods from 1870 until present time into account, a selection had to be made. The following cultural services

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as they originated and established in different historical periods were examined in more detail:

- Nature-based tourism and local recreation
- Landscape aesthetics and inspiration by ecosystems and coastal landscapes
- Extracurricular knowledge systems related to nature
- Cultural heritage and symbolic meaning of nature
- Natural heritage
- Regional identity and homeland

Cultural services differ from the other ecosystem service categories primarily in that the opportunities for them are provided by ecosystems (and landscapes), but the services themselves only emerge through the interplay with human cultural perceptions and practices. Cultural services are not just "delivered" by nature (see Chap. 5.3). Cultural traditions being full of values are always presupposed in those services. Cultural traditions resonate with beneficial flows from ecosystems and such resonance constitutes cultural services. For this reason, some scholars argue that cultural services should not be treated as ecosystem services in the strict sense, because they do not arise from purely ecological interactions and often cannot be traced back to specific ecosystem processes and components. Nevertheless, we have studied cultural services in the project in order to shed light on all aspects that ecosystems and landscapes contribute to human well-being. Moreover, cultural services are often the ecosystem services that are most directly experienced and intuitively valued by people (as opposed primarily to regulatory services), although not necessarily under the term "ecosystem service". The previous language games by which such benefits have been articulated in the past do not yet entail "cultural ecosystem services". Therefore, we had to do translations into this terminology.

"Cultural ecosystem services" are strongly influenced by human perceptions and valuations, and are generally difficult to be expressed or measured in mere numbers or monetary values. This is often presented as a problem in the literature, but can also be an opportunity to incorporate other bodies of knowledge into the assessment, which we will present in the results section. Although a compilation of quantitative data is in principle possible for some services (for example, via statistics on nature tourism, the sizes of nature reserves or willingness-to-pay analyses), the qualitative dimensions of cultural services are more meaningful with regard to a reflective ethical evaluation. We would therefore like to focus our contribution on this scope of information which is often available in written form and enables the description and understanding of contexts. Our methods stem from cultural studies, as hermeneutical interpretation of different historical sources. The samplings of the sources and documents were performed in different archives and libraries from 2016–2019. These sources and documents are (historical) travelogues, artworks, and their history of origin, postcards, advertisements, photographs, paintings, survey results, texts of early local history and nature conservation, literature on acceptance problems of nature conservation after German reunification, among others. We have assessed our sources, records, and archives by the lens of cultural services and eudaimonistic values. This sub-chapter cannot document all these sources. It only presents some general findings and illustrates them by selected pictures.

22.2 Findings

Using two more detailed examples, namely tourism as well as landscape aesthetics, we will present some results of the ethical assessment of cultural ecosystem services in the following. Subsequently, we briefly summarize some results on the other cultural services.

22.2.1 Example 1: Tourism

From an environmental ethics perspective, we are primarily interested in the different motivations that have led tourists to spend their vacations on the German Baltic Sea coast over the past 140 years or so, and the conflicts that can arise between different tourist groups or between tourists and local residents as a result. (Our investigation focuses on the coastal zones which belong to today's Germany, but one should not forget that there were tourism and artistic colonies all along the coastlines of Pomerania, Eastern Prussia, and the Memel region ("Kurische Nehrung").)

Motives are practical reasons to act. They are not in themselves sufficient to act, but in combination with favourable boundary conditions (leisure time, transport infrastructure, purchase power), a motive stimulates a course of action. The boundary conditions improved during the nineteenth century for more wealthy social groups in Germany. The following central motivations related to coastal ecosystems have been identified:

– Sea air/water for health promotion
– Bathing life and socializing (independent of health aspects)
– Water sports
– Artistic inspiration
– Hiking trails
– Recreation from city life
– Nature experience

Some of these areas of tourism go back further than the period under study (for example, the first seaside resort on the German Baltic coast was founded in 1793), but they are still to be found today in varying degrees of importance and, according to surveys, still play an important role in people's choice of vacation region.

On the one hand, there is a certain continuity in these motivations and value references, but on the other hand, they are always subject to strong change, as can be seen in the example of bathing vacations. Not only the bathing fashion, but also the access to the water (first from the bathing cart, later from the jetty in the bathing establishment, finally directly from the beach) has changed a lot over the years and in the face of the prevailing ideas of propriety. There was rapid cultural change with respect to dressing codes and the right to undress on the beach. Nudism became prominent in Germany since 1900 (see Andritzky and Rautenberg 1989). Coastal zones became prominent locations for nudism also in the German Democratic Republic.

Today, many elements of historic bathing life are linked to entirely new values (for example, bathing carts as wedding venues) or primarily serve to transport the beach feeling to areas far from the beach (for example, beach chairs in downtown Kiel).

The "fishermens' life" also forms an important attraction factor for tourists. Although the economic importance of fishing on the German Baltic coast has declined sharply in recent decades, it nevertheless (or perhaps because of this) has great cultural significance. In recent years, numerous fishermen's sculptures have been erected in coastal communities, and various more or less traditional fishermen's festivals and the ubiquitous fish sandwich stands also attract visitors. The sight of fishing boats and "real" fishing harbours, such as those still to be found in Freest or on the Holm in Schleswig, is also particularly valued.

Although fisheries have lost significance for local and regional economies, they are valued as a kind of local tradition and appreciated by nostalgia. The provisioning services of fisheries transforms into the cultural "service" of small harbours and the "flair" of artisan fisheries.

22.2.2 Example 2: Landscape Aesthetics and Inspiration

For centuries, the German Baltic Sea coast has attracted artists who have incorporated their impressions into a wide variety of art forms, such as paintings, sculptures, poems, musical pieces, and so on. In terms of capturing these inspirations and their effect on the human imagination (imagination) as an ecosystem service, it is interesting to investigate which components of ecosystems actually served the artists as the basis for their works.

We will illustrate the choice of motifs by painters in the period under study by using the paintings of the early phase of the Ahrenshoop artists' colony as an example. This artists' colony with many female painters ("Malweiber") was founded in 1892, as artists were attracted by the special light on the coast, the beauty of the landscape and the "simple" life in the village. The following elements of the coastal landscape were the main inspiration for the painters' works:

– The Baltic Sea near the beach (e.g. Louis Douzette, An der Ostsee, 1898)
– Dunes (e.g. Georg Kaulbach, Stranddüne Ahrenshoop, 1920)
– Trees being shaped by coastal winds ("Windflüchter") (e.g. Paul Müller-Kaempff, Kiefern am Strand, c. 1910)
– Bodden waters with/without boats (e.g. Friedrich Wachenhusen, Evening on the Bodden, around 1905)

– Thatched houses (e.g. Elisabeth von Eicken, Das Dornenhaus in Winter Tauschnee, 1890)
– Bodden meadows with/without cattle (e.g. Fritz Grebe, Bodden meadows, around 1895)
– Harbor scenes (e.g. Carl Malchin, Harbor of Wustrow with Laundresses, c. 1890)
– The steep coast (e.g. Dora Koch-Stetter, Hohes Ufer, around 1913)

Artists became residents at the coastal zones over months and explored peripherical regions. An exhibition in the Pomerian Gallery at Greifswald showed the paintings of Max Pechstein and Karl Schmidt-Rottluff in a comparative way. Both painters were in search for "original" ("urwüchsige") landscapes being undisturbed by industries. The cultural service of coastal landscapes is its "otherness", as compared to urban life in industrial areas. Ironically, such otherness diminishes with mass tourism, but as long as some otherness remains, the region remains attractive for tourists.

Overall, it is noticeable that the artists placed a special focus on different light moods and atmospheres as well as on weather and seasonal changes, as shown, for example, in the images by Müller-Kaempff and von Eicken. Their interest focused primarily on landscape elements that still arouse the enthusiasm of tourists (and local residents) today, which is reflected, for example, in the advertising photos of tourism websites and on portals for amateur photographers. Some might argue that the aesthetic appreciation of artists has been trickled down to popular culture. But one can also argue that the coastal environments have both been appreciated by popular culture and have inspired artists. Visualization of sites and situations that bring about cultural services are to be found in postcards, advertisements, photography, and works of art.

Moods and atmospheres, as we find it in many paintings, are a common topic for environmental phenomenology and aesthetics which deserves closer attention in cultural studies. Even if the ontological status of atmospheres remains dubious (Böhme: "Halbdinge"), atmospheres can be beneficial to individuals. Atmospheres are not about nature as such, but nature at dawn, on foggy afternoon, on hot summer day, at sunset, on stormy weather, etc. Atmospheres bring about embodied moods, and moods may have been transformative to sensibilities and attitudes. Atmospheres have eudaimonic recreational value, as they make one feel more relaxed and provoke a feeling of unity between bodily and mental states. Stormy coastal weather, for instance, provokes physical exercise and a sharp and clear spiritual mood (Fig. 22.1).

22.2.3 Other Cultural Services in Brief

With regard to extracurricular knowledge systems, we have primarily focused on nature trails and nature education offerings (guided tours, hands-on activities, etc.) in the study area. While the first German nature trail was opened as early as 1930, more and more nature experience trails have been established recently, which are intended not only to inform visitors but also to offer them direct experiences. We included

Fig. 22.1 (**a**) Paul Müller-Kaempff, Pines on the Beach, c. 1910. (**b**) Elisabeth von Eicken, The thorn house in winter exchange snow, 1890

trails in our study because trails lead to specific locations (as sightseeing spots) where some cultural services, as beautiful sceneries, become "highlighted". In the area of the German Baltic Sea coast, this is expressed, for example, in the climate and coastal experience path in Laboe or the experience path "De Lütt Küst" at the national park house on Hiddensee. Some recent trails have been inspired by environmental ethics (eudaimonic values, virtues, biophilia) and have been arranged accordingly. The "Pathway of Leisure and Recognition" ("Pfad der Muße und Erkenntnis", see Deickert 2013) near Lauterbach on the island of Rügen within the coastal forest reserve of the Goor combines scientific information on forests and coastal dynamics with thoughts on beauty, coming closer to nature, tranquility, calmness, and deceleration. The history of the Goor ist given by Jeschke, Knapp (2007). The entire triangle of the park of Putbus, the Goor forest, and the Vilm Island (Buske 1994) is a "hot spot" for cultural ecosystem services. The Goor trail has been composed as a "medley" for such services.

In terms of cultural heritage with a coastal connection, there are on the one hand (in addition to the art forms of painting, poetry, literature, music, etc., treated separately as an ecosystem service "inspiration") the arts and crafts (Freest fishing carpets, amber jewellery, thatched roofs, and much more), and on the other hand various customs and traditions such as barrel cutting (primarily on the Darß) or the herring bet in Kappeln. Many of these cultural practices, which often originated in connection with coastal fishing, are nowadays an integral part of the tourism portfolio of the regions concerned. The tension today is between an eventful and a more recreational mode of tourism.

When considering natural heritage, an increased focus on certain characteristic animal species is noticeable, especially in recent decades. In the National Park Vorpommersche Boddenlandschaft, for example, these are the migrating cranes and the red deer, furthermore seabirds play an important role. The spectacular and joyful scenery of several thousand cranes flying to the Southern spots ("Gellen") of the island of Hiddensee attract tourists as well as local people.

The German Baltic Sea coast also makes an important contribution to the regional identity of the residents, which is of course closely linked to many aspects already mentioned, such as coastal customs and fishing. In this context, texts of the Heimatschutz from the turn of the nineteenth to the twentieth century are particularly informative, for example, about the beginnings of the preservation of natural monuments and the preservation of the native characteristics of dunes, beaches, and kink landscapes. Among contemporary locals, there is a strong sense of belonging although many young people left (or had to leave) Mecklenburg-Vorpommern after German unification in the 1990ies due to high unemployment rates. Meanwhile, the coastal zones have become destinations of gentrification (see Schmidt 2017) because more wealthy people move to these areas (artists, digital professional, retired persons, descendants of former feudal landlords).

22.3 Historical Framings

The 'use' and appreciation of the cultural achievements of the German Baltic coast has been subject to numerous changes over the past 140 years. These changes in customs and perceptions always reflect the prevailing moral concepts, the political frame, and a change in the underlying values, for example, in relation to the bathing industry briefly described above or to the changing importance of fishing on the German Baltic coast. It is also evident that cultural services are often closely linked to each other (e.g. tourism and landscape aesthetics) or to other ecosystem services (e.g. fisheries as a provisioning service as well as an inspiration for cultural practices).

In addition, embeddedness in social and political contexts and frames should not be neglected. For example, the artist colonies on Hiddensee and Darß offered women artists opportunities toward the end of the nineteenth century that they did not have in the male-dominated art world of the urban academies. Thus, the Baltic Sea became a locus of artistic and poietic emancipation of women.

There was a first wave of proto-ecological movements in the years before the First World War. Economic prosperity of the middle- and upper-class motivated persons to seek refuge from industrial and commercial life in some peripherical region. Improved transport systems opened new opportunities to leave town and visit the countryside and the coastlines. Tourisms became a new and often flourishing business model. There were fashionable and distinguished destinations as the region Heiligendamm and Bad Doberan as well as the famous chalk cliffs on Rügen.

In the 1920s and 1930s, however, Jews were increasingly excluded from community life in the seaside resorts; at the same time, a huge vacation resort for up to 20,000 Aryan vacationers was to be built in Prora (Island of Rügen) under the concept of "*Kraft durch Freude*" which had some conceptual origins in the Fascist Italian "*Dopolavore"* (see Liebscher 2009). The naturalistic doctrine of the regime demanded a healthy population performing physical exercise and forming a collective body. The Prora buildings demonstrate how beach life might be politicized.

In GDR times, the beaches of Mecklenburg-Vorpommern had an ambivalent meaning, on the one hand as a place of longing, on the other hand as a place of confrontation with armed border guards. People had to leave the beaches at sunset and there were control stations to prevent people escaping GDR by boat. On the other hand, GDR books praised the coastal zones and islands as Hiddensee as real property of the people instead of privileges of the rich tourists (Wurst 1968).

Since 1990, two national parks (Boddenlandschaft, Jasmund), a biosphere reserve on Rügen, and many nature reserves have been established. The National Park Programme of GDR being fostered by a small group of persons was finally successful (Succow et al. 2001). Implementation of National Park programmes, however, were accompanied by local protest since local people fearing nature conservation might hamper economic development. Meanwhile, however, most persons have arranged with the situation and realize them as being profitable in the longer run. Despite some convergence, the tourist infrastructure differs from the more FDR style of Schleswig-Holstein and the post-GDR style in Mecklenburg-Vorpommern.

Particularly in light of the last point, recognition of cultural services or values offers opportunities for a broader rationale for conservation action in the context of the ecosystem services debate. Arguments about the fundamental reliance on regulatory and provisioning services can be juxtaposed with eudaimonistic arguments for maintaining these services through a closer look at cultural ecosystem services, since ecosystems as tangible landscapes also make an important contribution to successful, fulfilling human lives.

However, it is important to emphasize at this point the ambivalence of (especially) cultural services, as they can be perceived very differently by different people and can therefore easily conflict with each other as well as with other ecosystem services or political goals for action. Therefore, it is particularly important to disclose and discuss the underlying motivations, perceptions, and values in each individual case.

22.4 Conclusion

There is strong evidence in the sources and records that cultural ecosystem services are essential for appreciation of the Baltic coastlines since the origin of modern society in the nineteenth century. Despite much cultural change, such services do not diminish over time. They are enshrined in the cultural memory of coastal regions, as in local museums, and are actualized by tourism and its management. As our study strongly also indicate, cultural ecosystem services can be framed by competing political ideologies. As we suggest ethically, one should be aware of such framings.

To actualize cultural ecosystem services via tourism and to preserve the difference between coastal zones and urban areas, including the objective of nature conservation, will remain a task for coastal zone management in the twenty-first century, which should promote sustainable coastal lifestyles with undiminished ecosystem services.

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Economic Valuation of Cultural Ecosystem Services

23

Katharina Elisabeth Franck and Martin Benkenstein

Abstract

Conjoint Analysis (CA) originates from product design in business economics. By using a decompositional approach, part-worth utilities of single specifications of different characteristics can be derived from the total benefit of a product. In the conducted field study, the CA was applied to gain insights into tourist's and resident's perspectives on the valuation of cultural ecosystem services of the German Baltic Sea coast. Between May and July 2018, tourists and residents were asked in four locations to rate different specifications that describe characteristics of the coast in combination with a certain price level for a holiday stay/a home rental. The characteristics were chosen to be of strategic relevance when it comes to decision-making processes of stakeholders. Characteristics included "Coastal Infrastructure", "Water Quality", "Naturalness of the Beach", "Watersports" and a price. Breaking down the interviewees willingness to pay for a certain combination of specifications, a prioritization of the latter as well as an economic evaluation of the single specification were derived. The results show that valuation of cultural ecosystem services is possible using CA. The results show as well that the economic value of specific characteristics differs between different stakeholders.

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23.1 Introduction

Most of Germany's Baltic coast is not only a place for coastal residents, but also an attractive holiday destination. This makes the Baltic Sea coast a tourist destination that requires a certain amount of management. Also residential Areas must get managed. In order to be able to carry out this coastal management duties in a future-oriented manner, decision-makers need to have knowledge of the economic benefits of individual ESS from the perspective of tourists and residents. Especially concerning the conflict between sustainable resource protection and economic benefit optimisation decision makers need reliable information about the value that can be attributed to individual ESS. In order to do justice to both residents and temporary guests and to uncover different preferences both target groups must be equally considered.

Compared to utility services or regulatory services, the assessment of cultural services required here is considerably more difficult, as they are subject to an individual subjective perception and evaluation. In the present study, such ESS are the focus of the analysis. The objectives are, on the one hand, to identify and apply a method that can be applied in the practice of cultural ESS management. On the other hand, the aim is to gain new insights from the results of the field study. Cultural ESS are chosen in such a way that the possibility of a direct transfer of the approach into practice is ensured if the applicability is demonstrated.

23.2 Theoretical Background

Research on the monetary valuation of ecosystem services (ESS) is already well advanced (Peterson and Sorg 1987; Boxall et al. 1996; Adamowicz et al. 1998; Pearce and Turner 1990; Faber et al. 2002; Brander et al. 2006; Alrikson and Öberg 2008; Sagebiel et al. 2016). Looking at the different areas of ecosystem services, market prices are used for utility services (Brander et al. 2006; Sagebiel et al. 2016) and cost-based methods for regulatory services (Sagebiel et al. 2016). An evaluation of cultural services is much more challenging because hedonic utility dimensions make monetization difficult.

The economic value of ecosystems can be divided into utility and non-benefit values (de Groot et al. 2010; Salem and Mercer 2012). The sum of the utility values and non-benefit values is referred to as Total Economic Value (TEV) (Peterson and Sorg 1987; Pearce and Turner 1990). This concept assumes that the total value is made up of individual partial values. It becomes challenging at the moment when the psychological, mental or emotional experience of an ESS is to be taken into account, as it is the case with cultural ESS. One of the analysis methods used in this field is hedonic pricing, which assumes that the benefits and valuation are reflected in the price (Sagebiel et al. 2016, p. 993). Both extrinsic and intrinsic values are therefore recorded. The travel cost method can be used to determine which recreational benefit is attributed to nature. This involves analyzing the effort required to reach a specific region. Both the hedonic pricing and the travel cost method proceed indirectly and count among the revealed preference methods. The stated preference methods include contingent valuation methods, choice experiments and benefit transfer (Sagebiel et al. 2016; Adamowicz et al. 1998). With these methods, the maximum willingness to pay for an improvement of a situation is queried directly, or several alternatives of states are presented for selection (Sagebiel et al. 2016; Adamowicz et al. 1998).

In order to gain a basic understanding of the perspective of residents and tourists on the value of certain aesthetic cultural ESS, stated preference methods must be used. However, according to Sagebiel et al. (2016), there is a great need to catch up on the use of these methods, especially in the area of Baltic Sea research (Sagebiel et al. 2016).

Of the stated preference methods, the conjoint analysis was selected to conduct the empirical study. The intention of the method is to gain a holistic view and assessment of an object, e.g. a specifically describes ESS by a single person. The method originates from socio-economic research, where it has mainly been used for marketing strategies (Schirpke et al. 2019). More precisely it aims at the quantification of the overall preference of a person based on underlying attributes, which leads to a quantitative measurement of the relative importance of certain attributes with respect to others (Rao 2014). For the evaluation, customers are asked about their preferences regarding different combinations of aesthetic cultural ESS features in connection with a specific price. From the total utility value, part-worth utility values can be derived for single features, which in turn can be converted into willingness to pay.

In the context of the present study, the tourist destination is represented by selected specifications of aesthetic ESS characteristics. The objective of the study is to determine the part-worth utility values of the single specifications within the ESS characteristic.

23.3 Implementation of the Study

The empirical conjoint study was conducted in May, June and July 2018 by the University of Rostock. Residents and tourists were asked by interviewers about their preferences regarding a number of ecosystem services in their residential or holiday environment. Residents were considered to be those participants who live anywhere within a maximum distance of 20 km from the Baltic Sea coast. Tourists included German citizens who had at least once spent a holiday on the German Baltic Sea coast or who were on holiday. Both target groups were surveyed equally at different locations along the German Baltic coast. In order to take into account potentially different preferences of respondents, four regions were surveyed in both lively as well as quieter, more natural locations. The following regions and places were selected: Schlei: Kappeln, Maasholm; Kiel Fjord: Friedrichsort, Laboe, Heikendorf; Darß: Zinst, Prerow, Born; Rügen: Binz, Mönchsgut, Lauterbach, Ummanz; Hiddensee.

Table 23.1 ESS characteristics and characteristic specifications

Coastal infrastructure	Water quality	Appearance of the beach	Watersports	Price [€] holiday p.p./week	Price [€] net cold rent
Harbour	Clear	Natural, sparsely visited	No watersports	450	420
Promenade or sea-bride	Murky	Cleared and levelled, much frequented	Non-motorized watersport	600	520
No infrastructure			Motorized and non-motorized watersport	850	640

Tourists and residents were asked about their preferences regarding selected characteristics of cultural ESS. In order to first limit the varying characteristics and to map them as precisely as possible, the participants of both target groups were placed in a consistent scenario by means of an introductory text, so that the focus was directed exclusively to the evaluation of the immediate surroundings of the hotel or place of residence.

The tourist scenario foresaw a seven-day holiday in a single room in a four-star hotel that met all the guest's requirements. Residents were put into the scenario of an apartment search of a three-room apartment with 75 square metres.

The ESS-scenarios considered in the conjoint study were constructed with regard to the following characteristics and their specifications (Table 23.1).

The specifications of the four characteristics representing cultural ESS were visualized by photos. A pre-test was carried out to ensure that the photos used were representative. The question was asked to what extent a photo shows a natural beach, for example. The respondents were representatives from the target groups of the main study. They voted on a scale from "I do not agree at all" (1) to "I agree completely" (5). The pre-test was conducted until all photos reached an average value of at least 4.0.

A selection of 16 combinations of the 10 specifications within the four ESS characteristics were used in the survey. Four photos representing a specification of the four characteristics were put together as an ESS-scenario. Using such photo-based scenarios, it was possible to measure the value of aesthetic cultural ESS. Figure 23.1 shows two examples how the ESS-scenarios was visualized in combination with the price information for the tourist target group.

During the survey, the photo-based method prevents that the respondents directly compared the different ESS-scenarios to be evaluated. During the survey, care was therefore taken to ensure that only one photo-based scenario including price information was looked at and that respondents did not page back and forth. By showing only one scenario at a time, it was possible to reduce method bias. For each of the scenarios, respondents indicated their preference on a scale from 1 (would not choose in any case) to 7 (would choose in any case). This indicates the probability

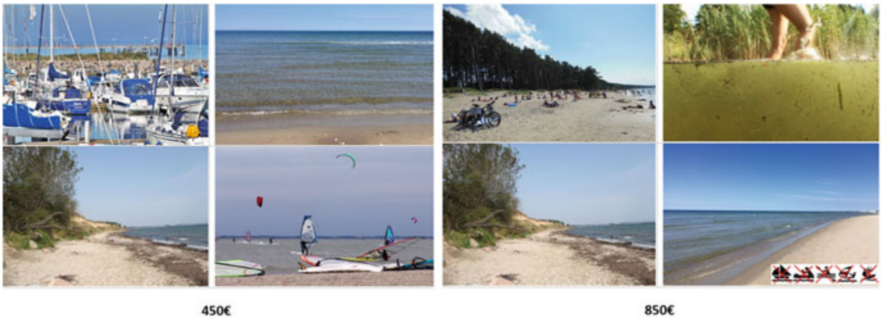

Fig. 23.1 Examples of ESS-scenarios in the tourist scenario

that they would be willing to pay a certain price for a certain ESS-Scenario. In this way interviewer-bias was tried to minimalize.

With regard to the prices used, respondents were also informed about the average market price according to the scenario for hotel or rent on the Baltic coast. Two further price categories were added to the survey, one lower and one higher price. One price category was then always combined with one of the characteristic specifications of the selected ESS features characterizing the environment.

A total of 236 tourists and 209 residents were interviewed. After correction of the data sets, a total of 233 tourists and 206 residents were included in the survey. In each survey, three incomplete questionnaires have to be excluded.

23.4 Results

The evaluation was performed using SPSS via the ORTHOPLAN function. ORTHOPLAN generates an orthogonal main-effects plan for a full-concept conjoint analysis. The resulting statistics provide, among other details, information on the utility values of single specification within a ESS characteristic (as relative information), as well as the corresponding prices. See Table 23.2 for tourists and Table 23.3 for residents.

The utility value–price ratio can be used to calculate the utility value corresponding to 10 € for the respective target group. For 10 € hotel costs, the utility value is 0.024 (1.099/45; 1.465/60; 2.075/85), for 10 € apartment rent, the utility value is 0.044 (1.833/42; 2.270/52; 2.793/64).

With this knowledge and with the help of the utility values given by SPSS for the selected specifications within the characteristics, the monetary value and the difference between these values can be determined for the specifications. The results of the calculations can be found in the section below.

Table 23.2 Part-worth utilities for specifications of characteristics given by tourists

		Estimated benefit	Standard error
Infrastructure	Harbour	−0.172	0.051
	Promenade/ sea-bridge	0.211	0.060
	No infrastructure	−0.039	0.060
Visual appearance of the beach	Natural, sparsely visited	0.640	0.038
	Cleared and levelled, much frequented	−0.640	0.038
Watersports	No watersport	0.127	0.051
	Non-motorized watersport	0.158	0.060
	Motorized and non-motorized watersport	−0.285	0.060
Water quality	Clear	−1.411	0.077
	Turbid/ murky	−2.823	0.153
Price	450 €	−1.099	0.106
	600 €	−1.465	0.141
	850 €	−2.075	0.199

Table 23.3 Part-worth utilities for specifications of characteristics given by residents

		Estimated benefit	Standard error
Infrastructure	Harbour	−0.174	0.062
	Promenade/ sea-bridge	0.16	0.073
	No infrastructure	0.014	0.073
Visual appearance of the beach	Natural, sparsely visited	0.483	0.047
	Cleared and levelled, much frequented	−0.483	0.047
Watersports	No watersport	−0.023	0.062
	Non-motorized watersport	0.133	0.073
	Motorized and non-motorized watersport	−0.11	0.073
Water quality	Clear	−0.954	0.094
	Turbid/ murky	−1.908	0.187
Price	420 €	−1.833	0.217
	520 €	−2.27	0.269
	640 €	−2.793	0.331

23.4.1 Results from the Survey of Tourists

In the following, the results of the survey of tourists are presented, structured according to the ESS characteristics considered.

When selecting different *coastal infrastructure* elements, respondents showed a clear preference for promenades and sea-bridges. Compared to no infrastructure, the willingness to pay increases by 104 € if a sea-bridge or promenade is available. In

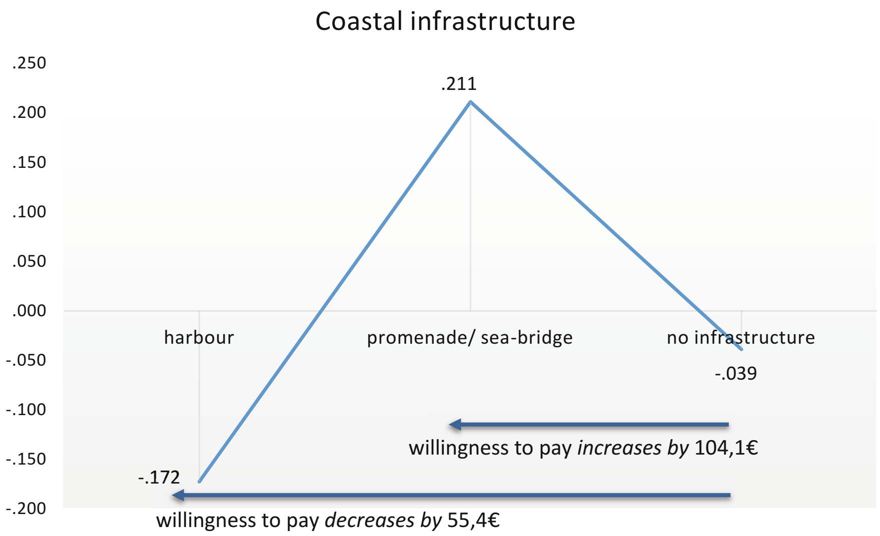

Fig. 23.2 Change in the value of coastal infrastructure from the perspective of tourists

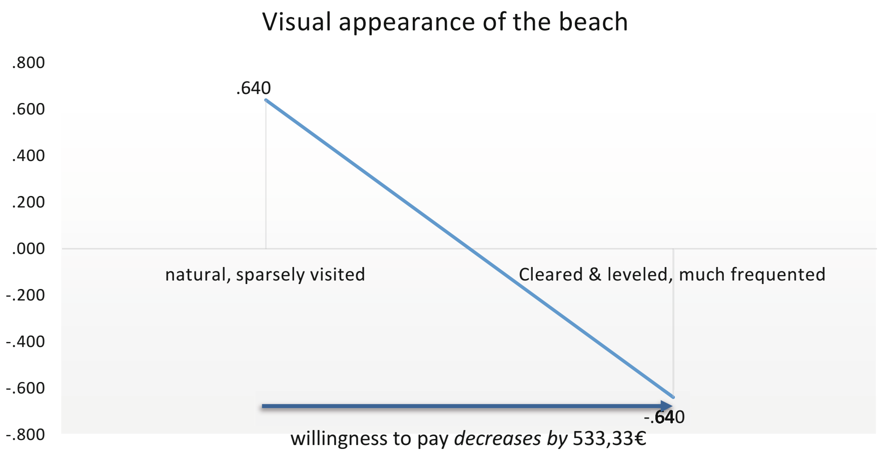

Fig. 23.3 Change in utility value through the appearance of the beach from the perspective of tourists

contrast, a harbour causes a price reduction of 55 € compared to no infrastructure (see Fig. 23.2).

With regard to the *appearance of the beach*, there is a clear preference for natural, less frequented beaches (see Fig. 23.3). Here the willingness to pay decreases by 533 € due to the appearance, if the beach is cleared, levelled and highly frequented.

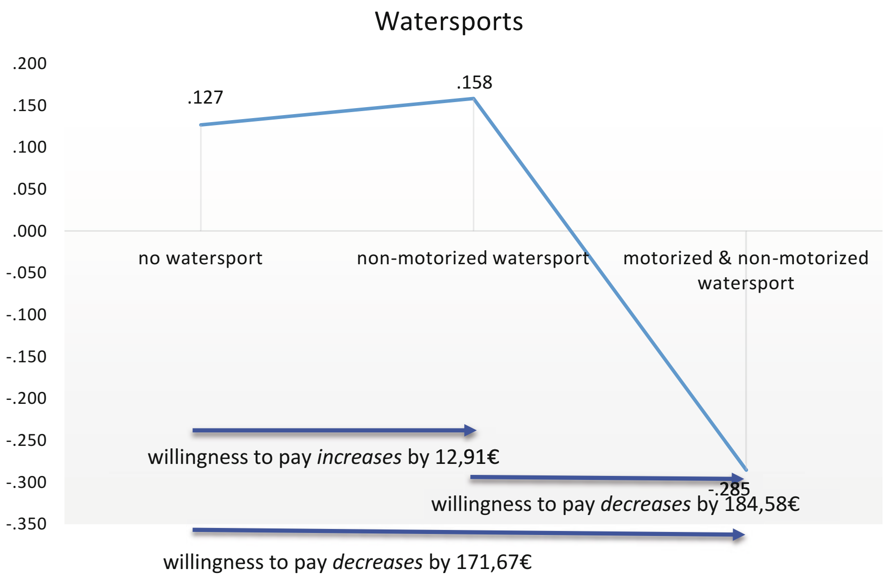

Fig. 23.4 Changes in the utility value of watersports from the perspective of tourists

Attentive readers will notice that this price change is higher than the cheapest holiday offer in the survey scenarios. For this reason, it should be pointed out that no linear relationship can be assumed here.

Finally, the example of *watersports* will be considered. The possibility of practising watersports is generally evaluated positively (Fig. 23.4). In contrast to a pronounced prohibition of watersports in the coastal shore area, the willingness to pay increases by 13 € if non-motorized watersports are possible. However, if motorized watersports are also possible, the willingness to pay decreases by 185 €. From no watersports to motorized and non-motorized watersports, there is a negative difference of 172 €.

23.4.2 Results from the Survey of Residents

In the following, the results of the survey of residents, structured according to the ESS characteristics, are presented.

From the perspective of the surveyed residents, a promenade or pier as a *coastal infrastructure* brings an increase in willingness to pay of 33.18€ in contrast to no construction. For a port, the willingness to pay falls sharply (42.72€), as it does from the perspective of tourists (Fig. 23.5).

The biggest change in the willingness to pay of both target groups can be seen in the *appearance of the beach*. For residents, this falls by €199 if the beach is cleared, levelled and highly frequented instead of natural and sparsely frequented (Fig. 23.6).

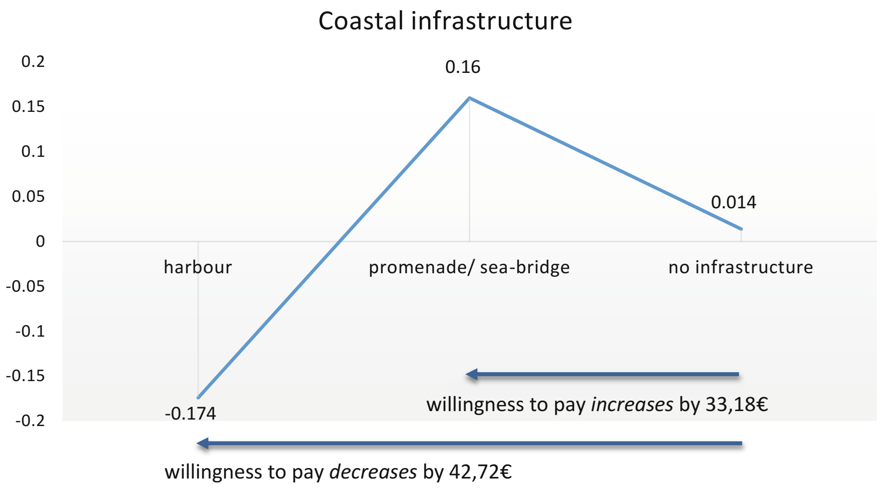

Fig. 23.5 Change in utility value through coastal infrastructure from the perspective of local residents

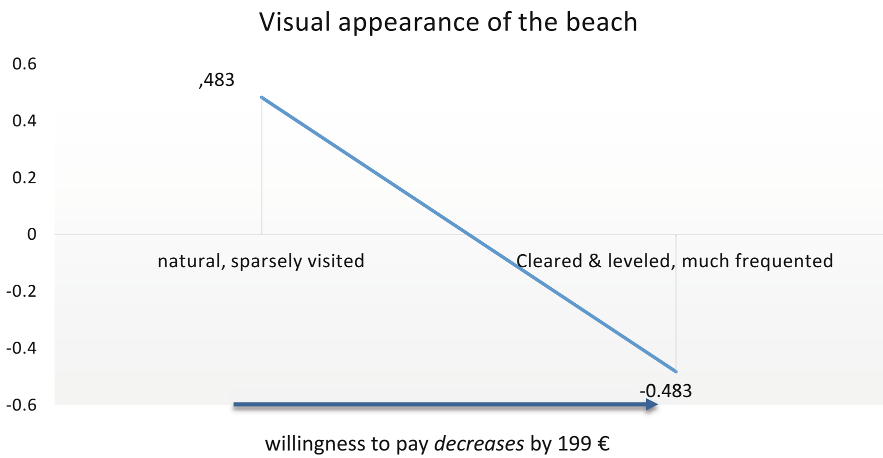

Fig. 23.6 Change in utility value through the visual appearance of the beach from the perspective of local residents

A similar dynamic as with tourists is evident in the question of the possibility of doing *watersports* (Fig. 23.7). While a prohibition of all watersports near the coast shows the lowest willingness to pay, this will increase by 35.45€ as soon as non-motorized watersports are possible. In the combination of motorized and non-motorized watersports, the willingness to pay is, in contrast to no watersports, lower by 19.77€.

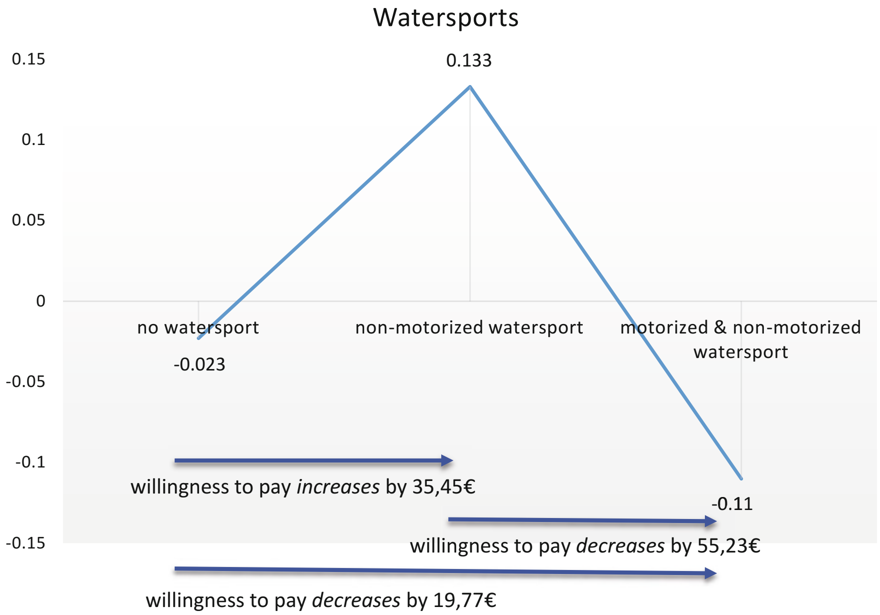

Fig. 23.7 Change in utility value through watersports from the perspective of local residents

23.5 Discussion

The results of the study allow conclusions to be drawn about the extent to which the willingness to pay changes as a result of changes in different specifications of ESS characteristics. In the present study, the preferences of both target groups, tourists and residents, are shown to be the same and only differ in the order of magnitude.

Although the application of conjoint analysis to ecosystem services leads to good results, there are some limitations, both methodically and in application, which require further research.

The specification of ESS tested here were selected for the concrete application in the field of destination management. The changeability through human intervention played a major role in this process. The exception to this is the evaluation of water turbidity, which, although it was found to be of utmost importance in the evaluation, is in fact not subject to linearity and can only be influenced by humans to a minimal degree. The application and testing of CA to other ESS is therefore reasonable.

A methodological limitation is clearly the linearity of value functions supposed by ORTHOPLAN. The results of this study show, that the value functions maybe non-linear. The beach can be empty, crowded or heavily frequented, but in between there are a thousand gradations whose differentiability is beyond analysis and is also subject to individual perception. This clearly indicates that the applicability of this measuring instrument can only lead to absolute monetary values for a specific ecosystem service to a very limited extent. What can be shown, however, is the trend as well as the order of magnitude in which the value of an ecosystem service changes due to specifications of ESS characteristics. The results presented can be interpreted in this sense.

The pictures had been tested in the run-up to the study in the format that participants rated on a 5-step scale to what extent they agreed to see, e.g. a beach with no infrastructure on the picture. This method of validation turned out to be unambiguous, but not unequivocal. If the question had been “What do you see?” the answer would probably not have been “a beach with no infrastructure”.

In order to avoid the purely visual interpretation of a stimulus, the interviewers used the possibility of additionally describing the presented stimulus in words if necessary (e.g. “The surroundings of the hotel show a beach with no infrastructure where watersports are prohibited. There is a harbour nearby and the water is murky.”) Unfortunately this option increased interviewer-bias.

To achieve balanced results, it is necessary to carry out the study at different times of the year. Different holiday intentions depending on the season, e.g. wellness and hiking holidays in winter as opposed to beach holidays in summer, would probably change the results significantly.

Great care should also be taken when selecting the survey locations in order to achieve a good mix of different target groups. A survey that is predominantly conducted in natural locations versus busy locations with promenades will result in corresponding preferences. The same applies to a survey conducted at popular watersports locations versus bathing beaches, where infrastructure and clean water are of prime importance.

The choice of the ESS characteristics was adapted to the conditions of the North German Baltic Sea coast in the survey carried out. The reader is asked to critically question whether the transfer to other coastal forms, regions and cultures is possible without restrictions.

23.6 Conclusion

Taking into account the limitation that specifications of the Characteristics of cultural ecosystem services are not subject to linearity as the applied methodology assumes, the study shows that the conjoint analysis is in general applicable to cultural ESS valuation. Alrikson and Öberg (2008) see the CA as the most suitable method for evaluating non-use values. The possibility of determining orders of magnitude of change in the willingness to pay for the specifications could be demonstrated, so that conjoint studies can be regarded as a promising tool for municipal decision-makers and can serve as an important basis for decisions in the management of tourist destinations as well as places of residence.

The experiences during the performance of the survey indicate that it is worthwhile to distinguish further groupings within the two target groups. The detailed investigation of the preferences of, for example different tourist target groups such as active holidaymakers or beach holidaymakers or the differentiation of different age groups of tenants can provide further helpful information. Age and family status might result in a significant difference in the preferences indicated. In the interests of the tourism economy, building planning and nature conservation, the application of the conjoint analysis will therefore be helpful and make it easier to go hand in hand in decision-making processes.

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Spatial Ecosystem Service Assessment Across the Land–Sea Interface

24

Johanna Schumacher, Sabine Bicking, Kai Ahrendt, Felix Müller, and Gerald Schernewski

Abstract

Various approaches to map and assess ecosystem services (ES) have emerged in the past decades. Yet, they are still mainly focused on terrestrial systems and comparable methods for marine systems are lagging behind. We present a joint spatial typology and ES classification for the German Baltic Sea to enable an ES assessment across the land–sea interface. Together, they built the basis for an expert-based assessment of ES potentials, and resulted in the first German Baltic Ecosystem Service Potential Matrix (Baltic ESP Matrix). We show its application for a spatial ES mapping across land and sea. Further, a complementary approach for an assessment along the direct coastline (i.e. where the water meets land) is presented. Systematic differences between terrestrial and marine systems, such as spatial and temporal variability of habitats, distinctness of boundaries and the 3-dimensional characteristic of aquatic systems, limit the comparability between both systems. In addition, subjectivity of experts resulting from differing knowledge, experiences, perceptions and traditions leads to dissimilar assessments of ES potentials at land and sea and affect experts' scores. To increase reliability of

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expert-based assessments, benchmark systems based on reference habitats could be used.

24.1 Introduction

The EU Biodiversity Strategy 2020 stresses the importance of ecosystems and their services and the need to maintain and restore them. Hence, it required all member states to map and assess the state of ecosystems and their services by 2014. For this activity, a joint spatial mapping approach is needed that allows an integrated assessment for land and sea. So far, most mapping approaches of ecosystem services (ES) have been developed for terrestrial systems, but are not directly transferable to marine systems (Burkhard et al. 2018). Lacking data and spatial delineations for marine systems, as well as differences in services and indicators, provide a major challenge. So far, the integration of existing approaches towards a joint spatial assessment across the land–sea interface has been hampered by these aspects.

To overcome this, a novel approach for a joint assessment scheme and spatial typology is needed, which considers the particularities of both systems and allows for a balanced spatial assessment for land and sea. Instead of starting from scratch, the assessment scheme and typology should build upon existing approaches, integrate available data and knowledge and international standards to increase acceptance. Furthermore, they should be based on current environmental policies to ensure practical relevance, applicability, data availability and regular updates.

In this context, our objectives are to present a spatial habitat typology and a linked assessment method suitable for a joint assessment across the land–sea interface. Exemplarily, we show their applications for visualizing the spatial distribution of ecosystem services in terrestrial and marine systems. In addition, we present a refined assessment for the coastline to emphasize specificities of this narrow ecotone and reflect our experience gathered during the development process.

24.2 Towards a Joint Spatial Typology for Land and Sea

Spatial subdivisions of landscapes into homogenous units form the basis for mapping ecosystems and the services they provide. This requires a spatially explicit typology for landscapes. The CORINE Land Cover (CLC) classification, which is based on satellite images, is widely accepted and provides a suitable background for many ecosystem service mapping approaches (Kandziora et al. 2013a; Schulp et al. 2014). One example is the matrix-based approach, that has been developed by Burkhard et al. (2012) and has been widely used for assessing ecosystem services (Campagne et al. 2020), for instance, in Northern Germany (Burkhard et al. 2014; Bicking et al. 2018). An ES matrix for terrestrial, coastal and marine ecosystems was published by Müller et al. (2020). However, it lacks spatially explicit units for marine systems, and is this not directly applicable for spatial assessment. Yet, it serves as a basis for our approach. While the CORINE classification meets the needs of terrestrial systems well, shortcomings exist for marine systems. For instance, inner and outer coastal waters of the German Baltic Sea area are represented by only two classes—'coastal lagoons' and 'sea and ocean'. Such differences in the spatial representation pose a major challenge for a joint assessment. Consequently, for marine systems, a comparable spatial typology is mostly missing. For its development, existing spatial units that are widely accepted and used in practice should be adapted. The Water Framework Directive (WFD) (2000/60/EC) and the Habitats Directive (HD) (92/43/EEC) can serve as a basis.

The WFD typology subdivides coastal and marine waters into comparable water bodies. It differentiates water bodies based on factors such as latitude, longitude, tidal range, and additional parameters such as depth, current velocity, residence time, and salinity. Therefore, water bodies, the management units of the WFD, which belong to the same surface water type, share many abiotic and biotic similarities. In this way, the WFD subdivides the entire seascape into comparable spatial units. This approach can be expanded to marine waters. However, being based on physico-chemical parameters of the water body, the WFD surface water types do not fully reflect the three-dimensional character of aquatic systems. Sediment and benthic habitat characteristics, which are lacking in the WFD surface water classification, are also important factors for the provision of ecosystem services.

The Habitats Directive includes a list of habitat types and species of community interest. It is closely related to the EUNIS classification (European Nature Information System—EEA 2017), which provides a hierarchical framework for coastal waters and marine habitats, and formed the basis for an EU-wide assessment of the distribution of marine ecosystem services (Galparsoro et al. 2012).

We built upon the classifications of the WFD and HD to derive a spatial typology for coastal and marine systems that complements the existing typology for terrestrial systems. We used the WFD typology of surface water types in German coastal waters of the Baltic, namely oligohaline inner coastal waters (B1), mesohaline inner coastal water (B2), mesohaline open coastal waters (B3) and meso-polyhaline open coastal waters, seasonally stratified (B4). This typology covers only coastal waters up to one nautical mile off the national baseline (a simplified coastline). We extended the WFD approach to marine surface waters so that all German territorial waters up to 12 nautical miles are covered. In line with the WFD classification, we separate coastal and marine surface waters by the 15 metres isobaths. Coastal waters comprise the shallow, light-penetrated and wave-influenced waters up to 15 m depth. Marine waters are deeper than 15 m. Therefore, they have a reduced vertical exchange and are potentially seasonally stratified, bearing the risk of hypoxia. Hence, all marine waters are included in the WFD surface water type B4.

Water body types B1 to B3 were further subdivided according to selected benthic habitat types, which were based on HD and HELCOM habitat and biotope types. This includes different sediment types, mud- and sandflats, vegetation and reefs. Also, the coastal ecosystem types, which are partly covered by CORINE, were redefined according to the HD and HELCOM coastal habitat types (e.g. 'sea dunes', 'sea cliffs, shingle and stony beaches' and 'salt marshes and salt meadows').

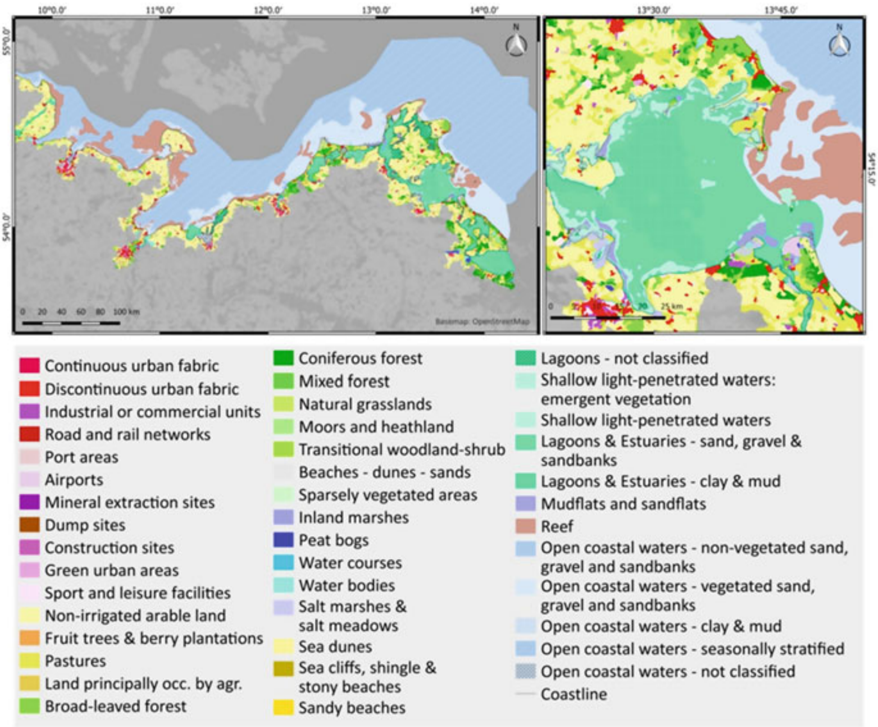

Fig. 24.1 Spatial ecosystem typology for a joint land–sea assessment along the German Baltic Sea (left) and exemplarily zoomed in on Greifswald Bay (right)

This resulted in a total of 14 habitat types for coastal ecosystems (5 types), inner coastal waters (4 types) and open coastal waters (5 types) defined for the German Baltic Sea area. These types were combined with the CORINE classes for the terrestrial area, which included settlement-related land cover (11 types), agro-ecosystems (4 types), forests (3 types), near-nature ecosystems (4 types), wetlands (2 types) and inland waters (3 types) (c.f. Fig. 24.1).

Geospatial data for the WFD water bodies and HD habitat and biotope types was readily available and could be easily obtained from public authorities. Data on macrophyte distributions was more difficult to obtain and was partly mapped using satellite imagery. Furthermore, less geospatial data on sediment distribution was available for inner coastal waters (e.g. Schlei and Darß-Zingst-Bodden Chain) than for outer coastal waters. All information was processed and joint into a single map (Fig. 24.1), which forms the basis for a joint assessment across the land–sea interface.

24.3 Towards a Joint Ecosystem Service Assessment Method

The common spatial typology and the resulting spatial separation of the seascape into discrete units build the framework for a joint ecosystem service assessment across the land–sea interface. The next step requires to define the scope of the assessment and to decide whether ecosystem service demand, potential or flow will be assessed. ES demand refers to the demand that is determined by individuals, interest groups or society in general. ES potential describes the capacity of an ecosystem (e.g. a mixed forest) to provide an ecosystem service (e.g. timber or wild food). It is defined as 'the hypothetical yield of selected ecosystem services' (Burkhard et al. 2012). The ES flow refers to the actual use of an ecosystem service in a specific areas and time and depends on the present ecosystem condition (Albert et al. 2016). Here, we focus on the ES potential that can serve as a baseline for subsequent assessments of ES demand or flow.

Following this, the ecosystem services that shall be assessed need to be determined. The Common International Classification of Ecosystem Services (CICES) provides an internationally accepted hierarchical framework that divides ecosystem services into provisioning, regulating and cultural services (Haines-Young and Potschin-Young 2018). We based our selected set of ecosystem services on CICES, but adjusted it in response to previous experiences gained during practical applications (e.g. Kandziora et al. 2013b; Inácio et al. 2018). Following the proposal by Müller (2005) and Müller and Burkhard (2012) a section for integrity attributes was added in order to also represent ecosystem conditions. Integrity indicators do not reflect ecosystem services as such, but provide information on the ecological state and quality of an ecosystem type. In total, our ecosystem services classification includes 6 integrity indicators, 14 provisioning services, 11 regulating services and 6 cultural services. The services are shown in detail in Fig. 24.2 (*y*-axis).

In the next step, the spatial typology and ecosystem services classification were combined in a joint matrix, called the German Baltic Ecosystem Service Potential Matrix (Baltic ESP Matrix). It shows terrestrial, coastal and marine land cover and habitat types on the *x*-axis and ecosystem services on the *y*-axis (Fig. 24.2). Using the matrix approach, the potential of each habitat type (*x*-axis) to provide a particular ecosystem service (or ecological integrity indicator, respectively) (*y*-axis) can be assessed.

We chose an expert-based scoring approach to carry out the assessment. Compared to scoring approaches based on models and statistical or field data, it is considered to include a higher degree of uncertainties and subjectivity. Advantages are that it is a relatively fast and simple method for assessing a large number of services for a variety of ecosystem types. As described by Müller et al. (2020) in detail, a scoring range from 0 to 100 is used to indicate the relative ecosystem service potential for each land cover or habitat type. The score 100 indicates the highest potential capacity of an ecosystem service. However, the Baltic ESP Matrix contains only values between 10 (very low ES potential) to 90 (very high ES potential). The value 5 indicates a provision that can be logically excluded (e.g. fish catches on arable land or in a forest). Allowing only scores between 10 and 90 can be

Land cover / Habitat types

		Settlement related CLC types (11)	Agroecosystem types (4)	Forest types (3)	Near-nature ecosystem types (4)	Wetland ecosystem types (2)	Inland waters (3)	Coastal ecosystem types (5)	Inner coastal waters (B1 & B2) (4)	Open coastal waters (B3 & B4) (5)
		Continuous urban fabric	Arable land non-irrigated	Coniferous forest	Natural grassland	Inland marshes	Running waters	Sea dunes	Shallow light-penetrated waters: submerged vegetation	Reef (1170)
Ecological Integrity	Abiotic heterogeneity	30	50	50	50	50	80	60	50	30
	Biodiversity	10	30	50	90	50	70	60	70	80
	Biotic water flows	10	50	70	60	70	10	30	5	5
	Exergy capture	10	90	90	70	70	30	30	70	60
	Reduction of nutrient loss	10	10	90	90	30	40	20	70	40
	Storage capacity	10	40	90	80	90	20	30	70	30
Provisioning services	Crops (human nutrition)	5	90	5	10	10	5	5	10	20
	Biomass for energy	10	90	10	10	10	5	5	20	20
	Crops (fodder), including aquaculture	5	90	5	10	10	5	5	10	20
	Livestock, including aquaculture	5	5	5	40	10	10	5	40	40
	Timber	10	5	90	5	10	5	5	5	5
	Fibers	5	10	10	10	20	10	5	30	30
	Wood fuel	10	5	90	5	5	5	5	5	
	Wild food	5	10	90	60	30	90	10	90	
	Fish and Seafood	5	5	5	5	5	90	5	80	90
	Flotsam	5	5	5	5	5	30	10	80	
	Ornamentals	5	20	10	10	10	40	30	30	
	Drinking water, service water and industrial	5	5	20	20	30	100	5	5	5
	Abiotic energy	5	50	5	20	10	80	5	20	30
	Minerals	5	5	5	5	5	50	90	5	
Regulating services	Groundwater recharge, water flow	5	50	90	80	50	90	40	5	
	Local climate regulation	10	40	90	20	70	70	20	30	60
	Global climate regulation	5	40	90	70	60	30	30	70	
	Flood protection	5	20	20	30	80	40	70	80	
	Air quality regulation	5	20	90	20	10	20	10	20	
	Erosion regulation, wind	5	30	90	80	20	5	10	5	
	Erosion regulation, water	30	30	90	80	30	20	5	70	60
	Nutrient regulation	5	30	80	50	70	30	20	90	
	Water purification	5	10	90	50	70	50	40	80	
	Pest and disease control	5	30	70	90	50	30	20	70	
	Pollination	5	30	40	60	50	90	20	80	
Cultural services	Recreation and tourism	10	40	70	50	60	80	70	40	80
	Landscape aesthetics + inspiration	10	50	70	60	60	80	70	60	70
	Knowledge systems	20	40	70	50	50	70	60	60	80
	Cultural heritage	40	50	60	60	50	80	70	80	80
	Regional identity	30	50	80	50	50	80	60	50	50
	Natural heritage	10	30	50	90	80	70	80	90	90

Fig. 24.2 Excerpt of the German Baltic Ecosystem Service Potential Matrix (Baltic ESP Matrix)—Land cover and habitat types are shown on the *y*-axis, ecosystem services (ES) on the *x*-axis. Expert-

considered artificial, but was chosen with regard to the inherent uncertainties of the approach. Moreover, it also allows to consider potential improvements or deterioration in case of scenario assessments (cf. Chap. 26). The value 5 was chosen to leave a minimal probability.

An internal expert working group provided a provisionally filled matrix, which was then sent to more than 100 external experts. They were asked to comment and propose alternative values. However, benchmarks for comparison were not provided. Comments by the external experts were considered in the preparation of the Baltic ESP Matrix. Finally, we reviewed it again to reduce mistakes and values resulting from misunderstandings, misperceptions and tried to adjust gradients between land and sea.

24.4 Mapping Ecosystem Services Across Land and Sea

With the filled matrix as a background, the spatial distribution of ecosystem services potentials can be easily visualized using GIS software. This can be done for single services, as it is exemplarily shown for the services ‘Global climate regulation’ and ‘Recreation and tourism’ in Fig. 24.3. The visualization allows a direct comparison between different services, but also between land and sea. Strong differences between inner and open coastal waters are shown for the ‘Global climate regulation potential’ (Fig. 24.3 top). Considering the high dynamics and exchange between inner and outer coastal waters, such strong differences between adjacent water bodies are debatable and could indicate a need for adjustments. Hence, by showing potential discrepancies, the visualization of ecosystem services potentials can support the evaluation of the ES potential matrix.

Variations between potentials for tourism and recreation are less pronounced for different land cover and habitat types (Fig. 24.3 bottom). Highest potentials are indicated for inner coastal waters, reef areas, rivers, lakes, and broad-leaved and mixed forests. Here, one has to keep in mind that ecosystem service potentials are reflected, but not the ecosystem service flows, which can vary widely due to environmental conditions, human demand or legal restrictions. Hence, spatial assessments of ES potentials do not reflect the actual ES provision, but they can serve as a baseline. In connection with assessments of ecosystem service flows or demands, they can help to identify areas of unsustainable use of ES (cf. Schröter et al. 2014; Baró et al. 2016) and thus support management and planning or the development of sustainability strategies.

ES potentials can also be shown for multiple ecosystem services on an aggregated level, as shown in Fig. 24.4 for provisioning, regulating and cultural services. Here, we used the average of all ecosystem services included in each section. A low

Fig. 24.2 (continued) based values in the inner cells indicate the potential for land cover/habitat type to provide the respective ES. The complete Baltic ESP Matrix can be found in Schumacher et al. (2021)

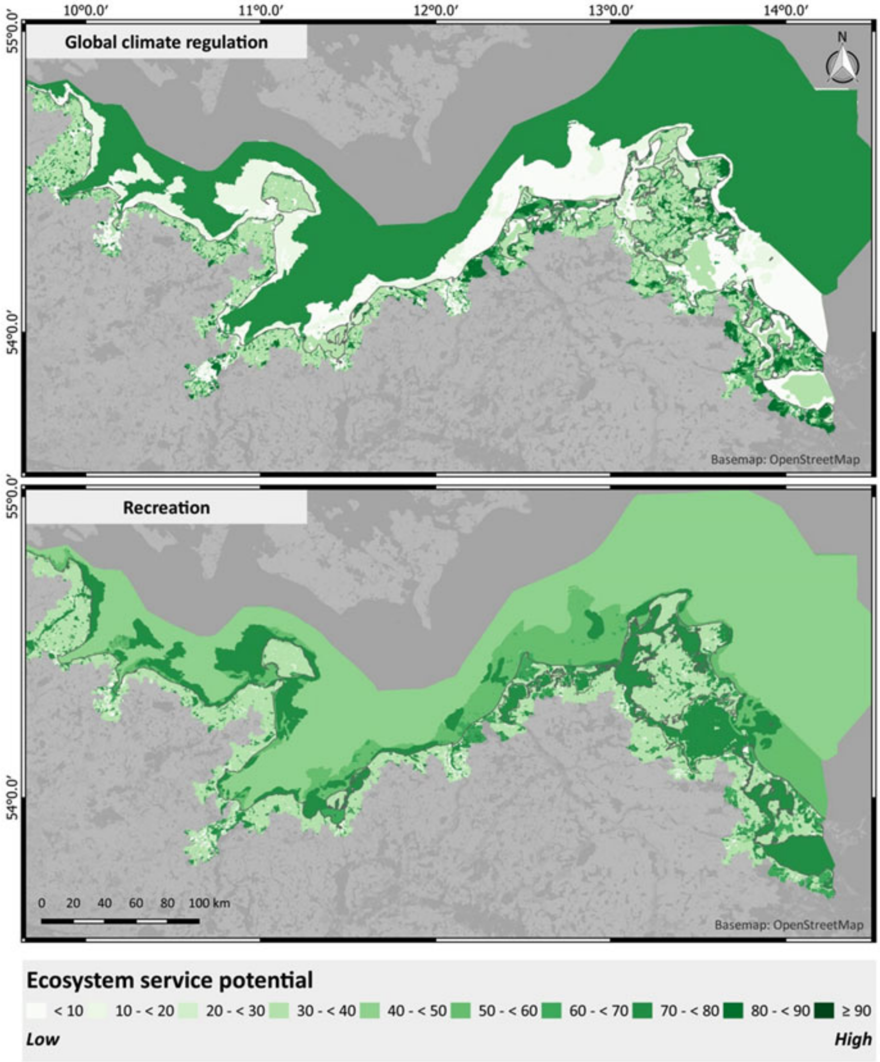

Fig. 24.3 Spatial distribution of the ecosystem service potentials for 'Global climate regulation' (top) and 'Tourism and recreation' (bottom) along the German Baltic Sea

potential is shown for provisioning services, particularly around urban areas. Values for regulating services vary widely among different habitat types. Lowest potentials are shown in urban areas, and highest potentials can be found along the coastal areas of Mecklenburg-Western Pomerania, especially along Darß-Zingst, and the islands of Rügen and Usedom. They can be ascribed to broad-leaved and mixed forests. Cultural ecosystem services, which include recreation, landscape aesthetics and heritage, show the highest potentials, especially for the habitat types 'lagoons and

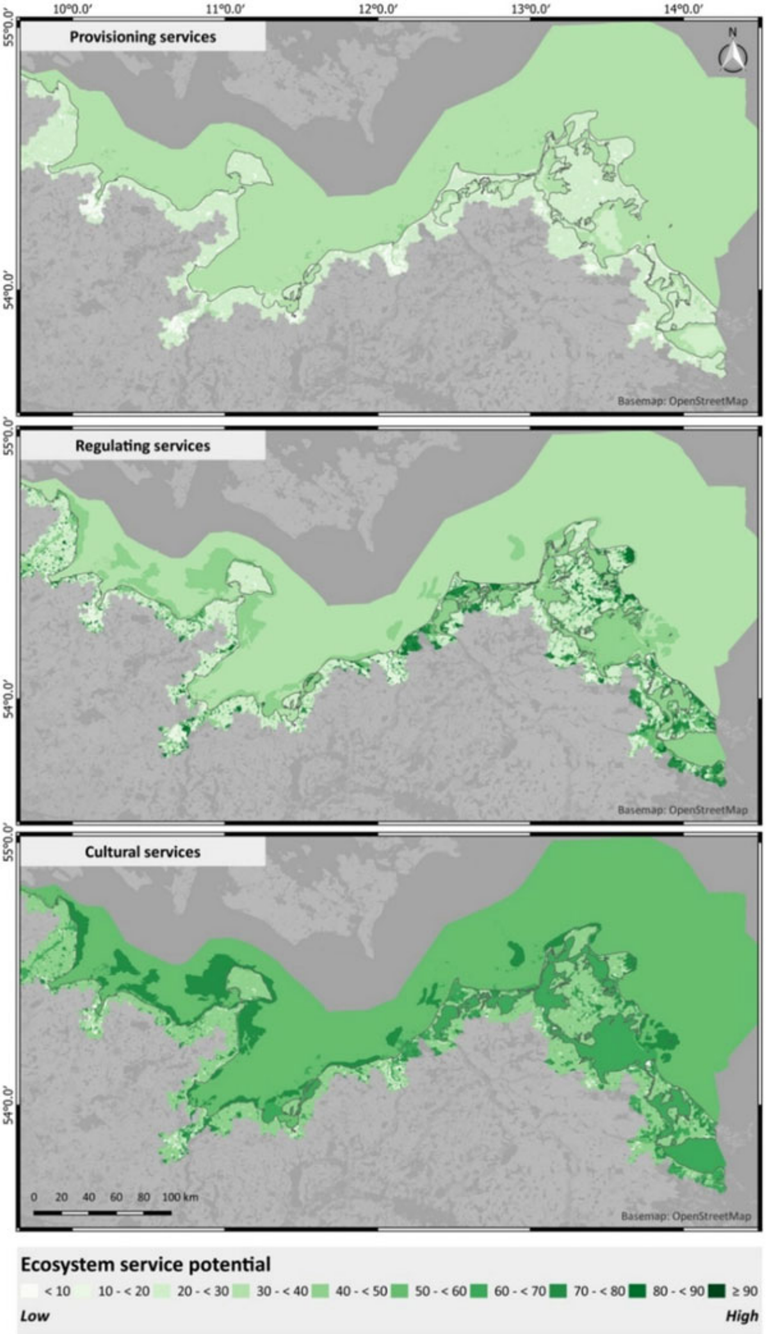

Fig. 24.4 Spatial distribution of the aggregated ecosystem service potentials for provisioning services (top), regulating services (middle) and cultural services (bottom) along the German Baltic Sea

estuaries' and 'open coastal waters with reefs'. This phenomenon has appeared in many instances of expert-based ecosystem service valuation, underlying the high significance of emotional and informational characteristics of the assessed habitats.

Assessing ecosystem services on an aggregated level can help to overcome weaknesses of single services. For instance, a direct comparison between land and sea of single provisioning services, such as fish and sea food or timber, is hardly applicable due to their restriction to either terrestrial or aquatic systems. In this case, the aggregated level enables a better comparability of both systems. Furthermore, the aggregated level gives an overview about the types of services that are provided in a region or by a particular ecosystem or habitat. However, when comparing the low potential of provisioning services with the high potential of cultural services, one has to keep in mind the peculiarities of both ES types. Provisioning services are often spatially mutually exclusive. For instance, a high potential provision of crops usually limits the provision of timber or livestock, as all services compete for the same space. In contrast, cultural services such as recreation and landscape aesthetics and inspiration can co-exist in the same space and can influence each other positively. Consequently, regulating services are prone to a lower potential provision on an aggregated level compared to cultural services. Yet, keeping this in mind, an assessment on an aggregated level supports the identification of dominating ecosystem service types and can serve as a basis for assessing ecosystem service synergies and trade-offs for planning purposes.

24.5 Assessing Ecosystem Services in the Coastal Zone

Due to its narrowness and high small-scale spatial variability, the coastal zone as the interface between land and sea has been hardly reflected in spatial ES assessments. Fixed scales that are often used in spatial assessments pose a problem and lead to a low representation of coastal areas. Yet, they are of high ecological as well as socio-economic importance. Hence, a refined and spatially more precise assessment approach for coastal zones that complements the joint assessment across land and sea seems to be reasonable and needed.

In this context, the aim of our work along the direct coastline (i.e. where water meets the land) was to develop a comprehensive GIS-based coastal classification scheme, which characterizes the coastal zone with its natural and socio-economic features and provides a basis for a respective ecosystem service assessment. For this purpose, a simple assessment methodology is used. It is based on the assumption that the coastline is made up of coastal segments with identical perpendicular conditions, land and seaward of the coastline. If one or more elements change, a new segment occurs. Hence, due to the high number of possible combinations, each segment could be unique. Through GIS software the littoral sections are segmented and classified according to 10 categories (Table 24.1). The segmentation and classification of the coastline are based on Google Earth maps and include a self-developed add-on for the GIS (Maptitude by Calipper Corp). Information on the ten classification categories is filled in for each segment.

Table 24.1 Categories for the segmentation of the coastline

Classification Categories	Examples
Dominant natural feature of the coastline	beach and dunes, beach, muddy coast/wetland, spit/beach ridge, cliffs (soft or rocky), barrier island, artificial coastline, atoll, delta, etc.
Substrate at the coastline	muddy (e.g. salt marsh, mangroves,), clastic sediments (compact, loose, gravel, sand), hard rocky coastline, artificial coastline, etc.
Dominant man-made features	harbour, coastal protection structures parallel to the shoreline (e.g. seawall, revetment, dike), or perpendicular to the shoreline (e.g.groins, jetties), residential and urban infrastructure, nourished/artificial beach, land reclamation structures, etc.
Additional man-made features	same as dominant man-made features if there are more than one man-made feature
Nearshore environment	lagoon, stream (mouth), spit, bay/inlet/gulf, longshore bars, tidal flats, mangroves, coral reefs, marshes, rocky platform, continuous slope, steep slope, land reclamation structures, breakwaters, etc.
Seaward environment	open sea, lagoon, delta, fjord, spit, estuary, etc.
Landward environment	dunes, marsh, barrier island, spit, cliffs (active/inactive), headlands, coastal plain, lagoon, continental plain, obscured by human development, etc.
Predominant land use	urban, rural (includes forestry and agriculture), industrial, transport, scattered settlement (villages), nature reserve, etc.
Other land use	same as predominant land use
Ecosystem services	Ecosystem service potentials for four classification categories (substrate at the coastline, dominant man-made features, nearshore environment, predominant land use) based on Müller et al. (2020).

For each segment, the ecosystem service potentials are indicated. They are generated based on the segments substrate, predominant man-made features, near-shore environment and predominant land use. Values for the ES potentials for the four segments were obtained from the ES matrix by Müller et al. (2020), which was also used as a basis for the Baltic ESP Matrix (above). They were additively aggregated into one single value, for each of the ecosystem services and integrity indicators and transferred into Arc Map. Using the Maptitude approach, the entire German Baltic Sea coastline was classified and ES potentials for each segment were indicated, as depicted in Fig. 24.5.

Ecosystem services potentials of different coastal segments can then be mapped. Comparisons between different parts of the coast can be made, e.g. comparing different regions, case studies, or inner and outer coastlines. This application is shown exemplarily on an aggregated level (for ecological integrity, provisioning, regulating, and cultural services and overall potentials) in Fig. 24.5 for the inner and outer coastline along the Schlei. Coastal segments with high overall ES potentials are shown along the inner coastline and in particular at the estuary mouth. Low ES potentials are shown in settlement areas, for instance, around the city of Eckernförde. Here, values for ecological integrity and regulating services are considerably lower in comparison to the coastal segments in the inner Schlei.

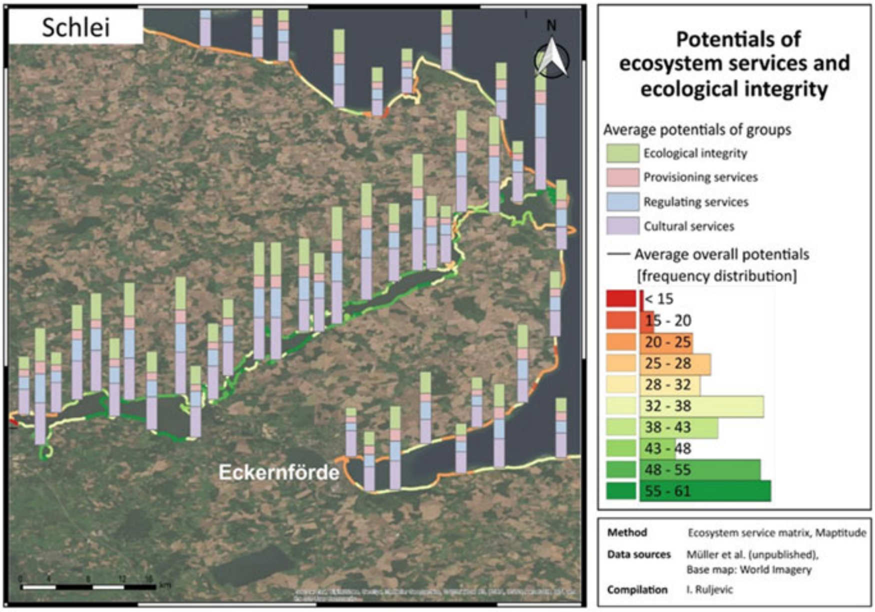

Fig. 24.5 Potentials of ecosystem services and ecological integrity (adapted from Ruljevic 2019)

Combining the Maptitude approach for the classification of the coastline with the ES potentials matrix allows a more precise and scale-independent ES assessment for the coastal zone, which is lacking in spatial ES assessments. Yet, being based on the same underlying matrix approach (i.e. Müller et al. 2020), results of the spatial ES assessment and the refined assessment for the coastline can complement each other, as they make use of the same set of ecosystem services and expert-based approach. Hence, they can be coupled in order to evaluate both the direct coastline and adjacent aquatic and terrestrial systems.

24.6 Lessons Learnt: The Spatial View Across the Land–Sea Interface

Disparities Between Land and Sea A typology covering terrestrial and aquatic habitats has to take into account the systematic differences between these systems. A meadow, for example forms a spatially well-defined habitat including soil, vegetation and the fauna adapted to this habitat. The counterpart, a shallow water seagrass meadow is spatially less well defined, because of its patchiness and it has to include the water body above. Beside the sediment, the water body defines growing conditions for seagrass, such as salinity, temperature or nutrients. Further, the water body itself forms a habitat with specific flora and fauna. But the plankton, living in the water column, is largely independent from the seagrass below and, because of currents, highly variable in time. Therefore, an aquatic typology has to cover three layers: the sediment, the benthos and the water body. Merging the typology of the EU Water Framework Directive with relevant habitats according to the Habitats Directive in the sea serves this purpose and allows a joint approach across the land–sea interface, but one has to keep in mind that the approaches for defining spatial units on land and in the sea differ.

The size of habitats across the land–sea interface is another aspect that differs between land and sea. On land, one square kilometre often shows a patchwork of different habitats, such as forest, meadow and cropland. Since land has well-defined owners and reflects human uses, these patch-works are spatially well defined and relatively stable in time. As a consequence, land cover patchiness is well reflected in available databases, such as CORINE. The sea is a common good and the habitats usually do not show spatial patterns resulting from human uses. Aquatic habitats are more natural, sometimes with a very small-scale spatial variability, sometimes showing a spatial uniformity. Small-scale spatial variability cannot be reflected in maps and only major changes in habitat controlling parameters allow separating habitats from each other. Further, aquatic habitats often face fast changes and spatial trans-allocations resulting from external events, such as storms. Last but not least the aquatic habitats are subject to a lack of data. Detailed land use data is available for centuries, because it always was important for humans. What is hidden under the sea surface meets less and only recent interest and cannot be explored easily at low costs, for example with remote sensing methods. The consequence of relative uniformity, spatio-temporal instability and a lack of data is that the spatial units in the sea are and have to be much larger compared to land as well as boundaries are partly artificial. For example using the 15 m isobaths to separate B3 and B4 is a simplification, because the depths of stratifications differ within the German Baltic Sea.

Subjectivity in Assessing Ecosystem Service Potentials Asking experts to estimate the potential of a habitat to deliver ecosystem services involves subjectivity. Scientific background, knowledge, experiences and perceptions play an important role and cause imbalances between the assessment of terrestrial and aquatic habitats. Already during childhood humans obtain detailed experiences and knowledge about terrestrial habitats and develop affections to flora and fauna. In childhood, aquatic habitats are mainly used for swimming and water sports, but without increasing the knowledge about benthic habitats much. In the Southern Baltic, aquatic flora and fauna are usually smaller and less visible and, by most, perceived as less attractive and interesting. Benthic habitats, hidden under the water surface, cannot be simply experienced by observation. As a consequence, even experts assume much higher ecosystem service potentials in terrestrial compared to aquatic systems, despite the fact that most terrestrial systems are hardly natural. The Baltic ESP Matrix tries to take this imbalance into account but despite that, the coniferous forest, which usually is an artificial plantation often covered with spruce, a non-native species, gets a high integrity value of 73 and an average ecosystems service potential score of 51. Vegetated lagoons and estuaries, the habitat with the highest ecosystem potentials in the sea, only show an integrity score of 56 (66 in case the integrity attribute 'Biotic water flows', which is not applicable in marine systems, is neglected) and a total average of 45.

The Terrestrial View on Ecosystem Services The concept of ecosystem service assessments has its most prominent origin in landscape ecology and has been intensively applied in terrestrial ecosystems. In comparison, applications in aquatic systems are lagging behind. The choice of ecosystem services which was adapted from Müller et al. (2020) still reflects this terrestrial background, for example the regulating services: groundwater recharge, flood protection or wind erosion regulation. This intrinsic focus on land is another reason for relatively low aggregated ecosystem potential scores in aquatic systems.

The Role of Traditions Ecosystem service assessment is an anthropocentric approach. It focusses on the benefits humans obtain from ecosystems. What humans perceive as benefit depends on the traditional uses of ecosystems and is not constant in time. Today in Europe, human uses focus on terrestrial systems and aquatic habitats play only a minor role. There are many reasons for this, for example accessibility, predictability of yields and depletion of aquatic resources. Today in Germany we observe an alienation with respect to coastal waters and the sea. Both are merely perceived as relevant with respect to selected cultural services, such as recreation and tourism or landscape aesthetics. In the early time of mankind, migration followed the coastlines, because the coastal zone provided a large variety of renewable food and enabled survival. A century ago in Germany, blue mussels were still intensively collected or cultivated on piles and fishing played an important role as a renewable source of protein. In eastern Asia, seafood is still very important. There, coastal waters are considered as indispensable for feeding the population (IPBES 2018). As a consequence, cultural events centre around coastal waters. This causes a very different perception of ecosystem service flows from and potentials of coastal waters in eastern Asia. The increasing demand for protein-rich food and feed and the potential of cost-effective mussel cultivations in coastal waters may move coastal waters in the focus of interest in Germany in the future again. However, the perception and appreciation of aquatic ecosystem services differ regionally and are changeable in time. The scores for the ecosystem service potentials in the Baltic ESP Matrix reflect the present tradition in Germany and this favours high scores for terrestrial habitats.

The Need for Reference Habitat Types One possibility to reduce the subjectivity in scoring ecosystem service potentials is to define a regional reference habitat. Preferably this should be the one with the highest overall ecosystem service potential, for example the mixed forest, which received the highest overall ES potential in the Baltic ESP Matrix. Experts would be asked to agree on scores for each ecosystem service for this habitat and all other habitats would be assessed relative to this reference habitat. To compare aquatic habitats with a terrestrial reference habitat would include all problems mentioned before. Therefore, an aquatic reference habitat could be defined, e.g. 'shallow light penetrated coastal waters with submerged vegetation'. The use of two separate benchmark systems would stabilize the scores for terrestrial and aquatic systems, but may cause a break between them. This could have effects on comparisons across the land–water interface. However, benchmark systems seem reasonable and should increase the reliability of the scores. As an alternative to regional benchmark systems, an absolute reference, valid all over the world, could be defined. Examples could be tropical rain forests and coral reefs.

24.7 Conclusions

With the Baltic ESP Matrix, we provide the first joint spatial typology and ecosystem service classification for an assessment of ecosystem services across land and sea for the German Baltic Sea area. The expert-based assessment of ecosystem service potentials and their spatial visualization illustrated the inherent differences between terrestrial and aquatic ecosystem types. Differences in habitat size and variability and the three-dimensional characteristics of aquatic systems, but also different levels of knowledge and available data complicate a joint assessment. Additionally, the cultural and professional background of involved experts influences the choice of ES potential values. Therefore, the valuations of marine and terrestrial service provisions are developed with a high degree of conceptual uncertainty, i.e. because the methods of potential determination are different in both spheres and also dependent on the experts involved. Consequently, the demonstrated joint assessment can be understood as a 'first set of hypotheses' integrating values of land and sea. They will be further examined through applications to concrete case studies and combined assessment of ES potentials and flows.

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25 Temporal Changes in Aquatic Ecosystem Services Provision: Approach and Examples

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Abstract

Marine ecosystem services are key elements to support human wellbeing. Yet, its provision is not static and may fluctuate over time as a response to ecological, social and political changes. We present a methodological approach and tool which is tailor-made to assess temporal changes in coastal and marine ecosystem services provision. The Marine Ecosystem Services Assessment Tool (MESAT) utilizes elements of the Water Framework Directive to spatially define the assessment units (water bodies) as well as temporal time slices (initial and present statuses). MESAT was applied to several water bodies in the southern Baltic Sea, including estuaries, coastal lagoons, bays and open waters. Changes in individual ecosystem services provision vary among water bodies. Nevertheless, general trends were possible to identify. The provision of the provisioning services has decreased over time, as response to anthropogenic pressures and ecological changes, and cultural services have increased significantly over time as a response to social and political changes. Regulating and Maintenance services showed no overall trends among the studied waterbodies. Assessing temporal changes can provide useful insights to managers and decision-makers by identifying trends and trade-offs between ecosystem services, as well as by highlight priority services to be restored in the future.

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25.1 Introduction

The potential benefits of ecosystem service assessments are well known, for example, to introduce an anthropocentric view on ecosystems, visualize the value of ecosystems for human wellbeing and support environmental protection. However, in practice ecosystem service assessments face many problems and challenges. This is especially true for coastal and marine waters (Liquete et al. 2013; Townsend et al. 2018). Compared to terrestrial ecosystems, for example, they do not have distinct boundaries. Water bodies are permanently moving and hardly stable in time and space. Further, ecosystems on the sea bottom, like mussel or seagrass beds, are not visible at all and subject to rapid changes after every storm. On the other hand, the ecology of marine habitats is relatively homogeneous and defined by major sediment characteristics and physico-chemical parameters. Another related major problem is the lack of data about aquatic systems. Insufficient data make absolute or monetary assessments difficult, and the results are associated with high uncertainties. High uncertainty limits the acceptance of the results and their practical relevance. Non-relevant results do not have an impact beyond the scientific world.

In coastal and marine waters, we have to use ecosystem service assessment approaches that meet these challenges and problems. They efficiently have to utilize the existing data and should build upon existing spatial unit definitions. Aquatic typologies should meet the requirements of coastal and marine policy to ensure an impact in the real world and should focus on relative changes in time to increase the reliability and usability of the results. Instead of absolute assessments, preferably two different states of an ecosystem in time are compared. For example, the pre-industrial state of an ecosystem, or a situation where an ecosystem was still in a desired good state serves as a baseline to compare the present situation or a future hypothetical state. Aims of this article are to describe a methodological approach to define suitable time slices and baselines, to exemplary compare and analyze temporal changes in coastal and marine water systems and to briefly discuss the relevance of this approach for environmental policy implementation.

25.2 EU Water Policy as a Framework

European coastal waters and marine ecosystems are under intensive and increasing human use and face ongoing degradation. As a consequence, the European Union (EU) water policies try to protect, restore and manage coastal and marine systems in a sustainable way. The Marine Strategy Framework Directive (MSFD, 2008/56/EC) establishes a framework for EU marine environmental policy and is implemented by existing legislation such as the Water Framework Directive (WFD, 2000/60/EC). The WFD classifies the ecological status of coastal waters based on biological quality elements, namely phytoplankton, macroalgae and angiosperms as well as benthic invertebrate fauna. The objective is to reach a "good status" in EU coastal waters, following a stepwise and guided process.

The definition of "good status" is based on reference conditions. According to the Common Implementation Strategy for the WFD (CIS 2003), reference conditions describe the biological quality elements that would exist with only very minor disturbance from human activities. They reflect a high ecological status. In practice, surface water ecosystems with only very minor disturbance hardly exist. Therefore, reference conditions are defined based on historical data and information, modelling and in case of need, expert judgement. In Germany, an official expert group decided to use the years around 1880 as reference conditions. With a spatially coupled, large-scale modelling approach this historic situation was reconstructed and spatially expanded to all German Baltic coastal and marine waters (Schernewski et al. 2015). The official expert group agreed that the targets for nitrogen, phosphorus and chlorophyll concentrations are defined by adding 50% to the reference concentrations for the "good ecological status". In practice for most German coastal waters, this reflects an ecological situation around the year 1960. As consequence, the WFD implementation process provides two baselines with concrete years for ecosystem service assessments, the desired good status (target value) around the year 1960 and the high ecological status (reference value) that was still present around 1880.

Coastal and marine waters differ and require specific reference and target values. To meet this requirement, the characterization and classification of all German Baltic coastal and marine waters was carried out, based on physico-chemical parameters (like depth, tidal range, salinity, temperature, turbidity, residence time, wave exposure and current velocities). The resulting typology subdivides the seascape into spatially defined ecological units with similar properties reference and target values. Coastal waters of one type are subdivided into smaller units, the water bodies, which form the management unit of the WFD. Altogether, the WFD provides a spatial subdivision of the seascape that is well suitable for an ecosystem service assessment because coastal waters belonging to the same type show many similarities with respect to ecological properties, structures and processes. This enables a certain spatial transfer (to comparable systems) of ecosystem service assessment results (Schernewski et al. 2019).

25.3 The Assessment Approach

To assess temporal changes of ecosystem services provision for coastal and marine water bodies, Inácio et al. (2018) developed a tailor-made methodological approach and tool, named Marine Ecosystem Services Assessment Tool (MESAT).

The tool adapts the Common International Classification of Ecosystem Services v4.3 (CICES) (Haines-Young and Potschin 2013) to coastal and marine waters. This classification divides ecosystem services into three sections: provisioning, regulating and maintenance and cultural. Further, these sections are hierarchically divided into divisions, groups and classes. In total 31 ecosystem services classes are considered, each represented by one or several indicators. A total of 54 indicators, adopted from the Project Mapping and Assessment of Ecosystems and their Services (MAES)

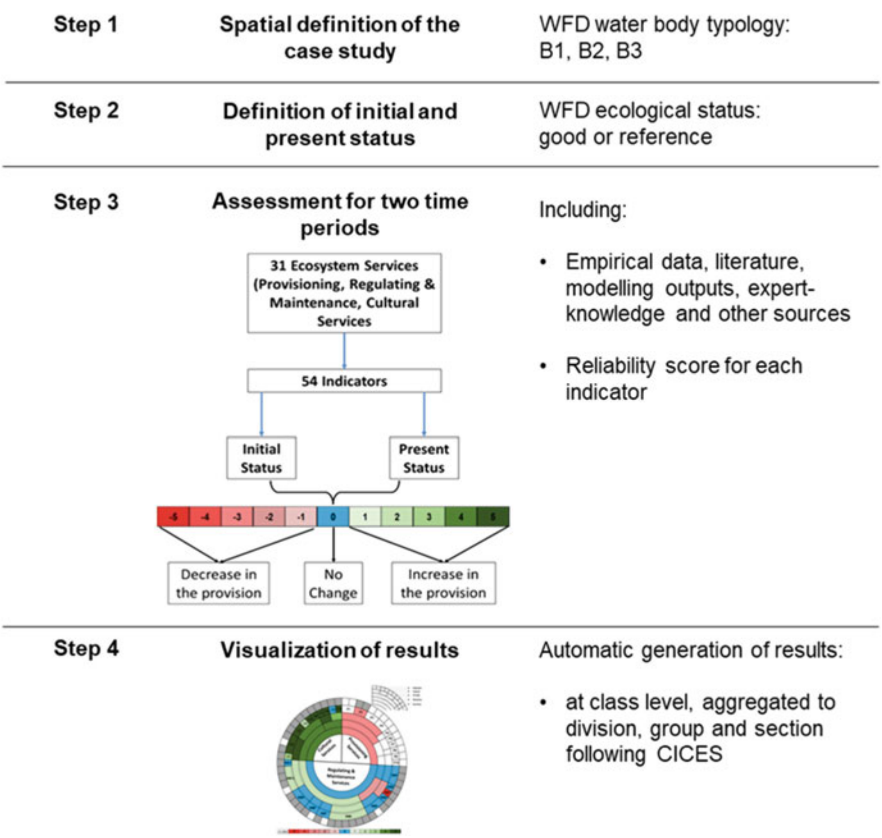

Fig. 25.1 The methodological steps which constitute the framework of MESAT—adapted from Inácio et al. (2018)

(Maes et al. 2016), as well as some newly developed ones, were assessed for the two time periods: an initial status and a present status. Overall, the assessment includes 10 provisioning services represented by 14 indicators, 11 regulating and maintenance services and 27 indicators, and 10 cultural services by 13 indicators. More details on these ecosystem services and their indicators are provided in Inácio et al. (2018).

The application procedure of MESAT consists of four steps (Fig. 25.1). For full description of each step see Inácio et al. (2018) and Schernewski et al. (2019).

New is that the tool utilizes two major aspects of the WFD, the typology that subdivides the seascape into discrete spatial units (Fig. 25.1—Step 1), the water bodies, as well as the temporal baselines (good and reference status).

In MESAT water bodies are compared at two different points in time (Fig. 25.1—Step 2). From a practical point of view, the reference or the "good ecological status" (hereafter initial status) compared to the present ecological status (hereafter present status) is of the highest interest. This is because it shows how the ecological

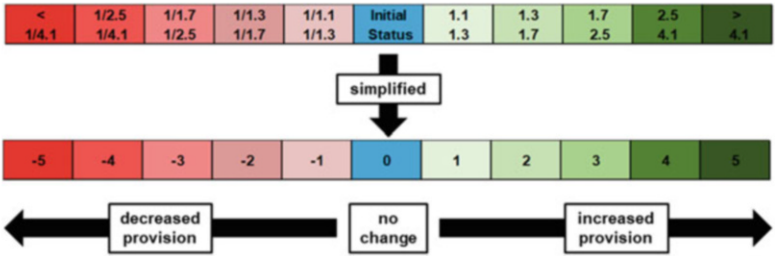

Fig. 25.2 Scale utilized in MESAT to allocate changes in the provision of ES comparing an initial with a present status

degradation of our water bodies during the eutrophication process affects their capacity to provide ecosystem services. However, other applications are possible as well, for example, comparing the present status with a hypothetical future status after the implementation of measures (Schernewski et al. 2019).

In step 3 (Fig. 25.1), data is collected for the identified services and indicators for the two time periods. Then, the comparative assessment is carried out allocating differences between indicator values or services in 11 classes, ranging from −5 to +5. 0 indicates no change in services provision, +5 a very strong increase over time and −5 a very strong decrease over time (Fig. 25.2). These classes are a simplification of the numerical fractional scale (Fig. 25.1—upper number) which delimitate each class boundary, based on the indicator values for the initial status. More details on MESAT are provided in Inácio et al. (2018).

25.4 Application Examples: Southern Baltic Coastal Waters

Altogether, the MESAT has been applied to six water bodies located in the Southern Baltic Sea. The case studies cover coastal lagoons, estuaries, bays and open coastal waters, as well as different environmental and anthropogenic conditions. Schernewski et al. (2019) apply the tool to the rural Schlei estuary and the urbanized Warnow estuary, both located in northern Germany. Inácio et al. (2018) applied the tool to the Szczecin Lagoon (Germany/Poland) and the Curonian Lagoon (Lithuania), two of the largest coastal lagoons in northern Europe. Inácio et al. (2019) applied MESAT to the Greifswald Bay (coastal bay) and Pomeranian Bay (open coastal waters), located in northeast Germany. Common to all case studies was the degradation of ecological status. Changes of biological quality elements and overall ecological status are fully described in the above-cited publications.

As the case studies are located in the same geographic region, the initial was the same for all case studies and defined as the time period around the 1960s (see Sect. 25.2) and the present status between 2010 and 2018.

25.4.1 Provisioning Ecosystem Services

In southern Baltic coastal waters, provisioning services mainly relate to the use of wild animals for nutrition and the use of fibres for construction materials.

An important provisioning service is the "use of fibres and other plant materials for direct use of processing" (Table 25.1, P6). In the southern Baltic Sea, there is a long tradition of utilizing reeds as material for roof building (Köbbing et al. 2013). However, the utilization of reeds as construction materials has decreased over the years. Reasons are the abandonment of the traditions, replacement of reed by other more durable and cheap material and strict environmental and safety regulations (Karstens et al. 2019).

Other provisioning services included the use of surface water for irrigation purposes, the use of macrophytes and mussels as agricultural fertilizers and the use of wild plants for nutritional purposes. Based on expert information, the provision of these services has, in general, decreased over time.

Traditionally related to fisheries, the provision of "wild animals and their outputs" (Table 25.1, P2) is the most important provisioning service in the southern Baltic coastal waters. At the time of the initial status (1960s), fisheries were an important, and for some case studies, the only provisioning service. In the Warnow, the

Table 25.1 Assessment of ES provision changes for Provisioning section at the class level for all MESAT case studies

	SE	WE	SL	GB	PB	CL
P1. Wild plants, algae and their outputs		-2				-2
P2. Wild animals and their outputs	3	-2	-3	-3	1	-3
P3. Animals from in situ aquaculture						-1
P4. Plants and algae from (…) aquaculture						
P5. Surface water for drinking purposes						
P6. Fibers & other materials from plants (…)	-5	0				0
P7. Materials from plants, algae (…)		0				-2
P8. Surface water for non-drinking purposes		0				-1
P9. Plant based resources						2
P10. Animal based resources						-2

SE Schlei Estuary, *WE* Warnow Estuary, *SL* Szczecin Lagoon, *GB* Greifswald Bay, *PB* Pomeranian Bay, *CL* Curonian Lagoon. Categories of change (CC) (based on the aggregation by reliability and for Curonian Lagoon aggregation by number): 0—no change; −1 to −5—decrease in services provision; 1 to 5—increase in services provision; grey colour—no data/not considered - adapted from Inácio et al. (2019)

urbanization of the estuary replaced fisheries by other more profitable economic activities. In general, the eutrophication process, which had its peak usually in the late 1980s, caused an increase in phytoplankton biomass and subsequently an increase in fish biomass. Further, one of the most important species for fishing in inner coastal waters, pikeperch, generally benefits from eutrophication and reduced water transparency. However, an increase in fisheries is only observed in the Schlei estuary and in the Pomeranian bay. In the open Pomeranian bay, it is likely that an increase in fish biomass allowed a slight increase in fisheries. Already in the 1960s, the Schlei was already heavily eutrophied and over-fished and the weak database suggests that fisheries simply recovered from a low level.

However, in most lagoons, such as Szczecin lagoon and Curonian lagoon, a decrease in fisheries is observed. Here, eutrophication caused a shift in fish species abundancy towards bony white-fish species. In the past, these planktivorous fish species had a market value, but consumer behaviour has changed. Today, only larger carnivorous fish species can be profitably sold, and several important species, such as eel, were largely lost. Reasons for the decrease in fish output are beside the lower value of most abundant species, overfishing of selected target species and international competition, causing reduced profitability and decline of fisheries in Baltic lagoons. Another reason is that benthic habitat destruction and the loss of important nursery grounds negatively affected the fish stocks. This is a consequence of reduced water transparency, resulting from eutrophication as well as mechanical destruction (fisheries, bathing, boating, anchoring). This plays an important role in Greifswald bay, which is a major nursery ground for herring. Poorer recruitment and overfishing caused a decline in fisheries. However, in open coastal waters and bays, where migrating fish species, such as herring, play an important role, fish stocks are also determined by external factors. In Greifswald bay, for example, it seems that reduced fish stocks are affected by climate change. Higher water temperatures in early spring seem to decouple the food-web, causing a lack of feed for juvenile fish.

Despite the decreasing trend, in the Schlei and Pomeranian bay, the provision of this service increased.

In general, provisioning services are declined in the southern Baltic Sea over the decades. Most important is the provision of fish. Its development over the decades is affected by eutrophication, habitat destruction and (over-) fishing, causing specific pattern for each water body. The results indicate that the provisioning services in the southern Baltic are not managed in a sustainable way.

25.4.2 Regulating and Maintenance Ecosystem Services

From the initial status (1960s), an increase in agriculture and population density as well as industrial and urban development led to problems associated with the increase of nutrient loads and other pollutants (e.g. Feibicke 2005; Radziejewska and Schernewski 2008), spatial occupation and in some cases destruction of natural areas including important habitats. The consequence was the degradation, to

Table 25.2 Assessment of ES provision changes for Regulating and Maintenance section at the class level for all MESAT case studies

	SE	WE	SL	GB	PB	CL
RM1. Filtration/sequestration/storage (...)	-2	-3	-2	0	-1	1
RM2. Dilution by (...) marine ecosystems	0	5	0	0	0	0
RM3. Mass stabilization and control (...)	3	-3	-4	-5	0	1
RM4. Buffering and attenuation of (...)		0	0	0		1
RM5. Flood protection	0	-1	-1	0	-1	0
RM6. Maintaining nursery populations (...)	2	0	-1	1	1	2
RM7. Pest and disease control	-1	1	-2	0	-2	0
RM8. Decomposition and fixing processes	0	1	0	0	0	-1
RM9. Chemical conditions of salt water	0	2	-1	0	0	1
RM10. Global climate regulation (...)	1	0	0	1	0	1
RM11. Micro and regional climate (...)	0	-1	0	0	0	1

SE Schlei Estuary, *WE* Warnow Estuary, *SL* Szczecin Lagoon, *GB* Greifswald Bay, *PB* Pomeranian Bay, *CL* Curonian Lagoon. Categories of change (CC) (based on the aggregation by reliability and for Curonian Lagoon aggregation by number): 0—no change; −1 to −5—decrease in services provision; 1 to 5—increase in services provision; grey colour—no data/not considered - adapted from Inácio et al. (2019)

different degrees, of water bodies in the southern Baltic, subsequently affecting its natural functioning and capacity to provide services.

One example is the ecosystem service "filtration, sequestration, storage and accumulation of toxics by ecosystems" (Table 25.2, RM1). In the past, the increasing nutrient loads to the water bodies led to an increase in the nitrogen fixation and burial of phosphorous and denitrification rates. While this increase in the filtration capacity is important, ultimately it does not arise from a propitious situation for human wellbeing. This is because an increase in nutrient leads to some extent to eutrophication processes. Therefore, the assessment resulted in the decrease of this service provision for Schlei estuary, Szczecin lagoon and the Warnow estuary. In Greifswald bay, despite the increase of nutrient loads, the open connection to the sea enables a high exchange rate, meaning that the changes which occurred are not that strong compared to closed systems (Valiela 2015). In the Curonian lagoon, the increase of services provision is associated with an artefact of a qualitative expert assessment, as Olenina and Olenin (2002) observe an ecological degradation of the system associated with the increasing nutrient loads.

Another service affected by ecological degradation is the "mass stabilization and control of erosion rates" (Table 25.2, RM3). The decrease of this service relates to the decrease of macrophyte coverage, observed in Greifswald bay by Munkes (2005), in the Szczecin Lagoon by Fenske (2002); and in the Warnow estuary by a decrease in the area of reed stands (Robbe et al. 2018). In the Schlei estuary, the increase of this service is related to the extension of reed belts surrounding the lagoon, which increased from the 1970s to today (LANU 1978), most probably due to nature conservation measures.

Climate change can also have a direct effect on the provision of ecosystem services, for example, "flood protection capacity" (Table 25.2, RM5). Climate change has a direct effect on sea level. Over the last century, sea-level rise has been observed in the Baltic Sea as well as an increase in the frequency of extreme weather events (rain) (Rutgersson et al. 2015). Thus, the decrease in provision of the flood protection service relates to the increase of the design-basis flood height and the increase of the significant wave height.

Another anthropogenic environmental problem influencing the provision of ecosystem services is the introduction of invasive alien species. The service "pest and disease control" (Table 25.2, RM7) is represented by the occurrence of harmful algal blooms and the presence of invasive species. In the Schlei and the Szczecin Lagoon, the provision of this service decreased, as justified by an increase in the number of invasive species, from 7 to 11 in the Schlei (Jaeckel 1962; Lackschewitz et al. 2015) and from 6 to 22 species observed in the Szczecin Lagoon (Gruszka 1999; AquaNIS 2016). In Greifswald Bay, no change in the number of invasive species, which today is about 31 species, was observed. In the Curonian Lagoon, however, the provision increased, however, this result addressed by experts does not go in line with the observed increase of invasive species in the past decades (Zaiko et al. 2007). In qualitative assessments, experts are required to refer to their knowledge on the ecological status of the case study for the two time periods, based on their perceptions. Hence, not all experts (general public) are familiar with scientific terms like ecological status and biologic quality elements. Therefore, the justification for this contradicting result can be either (1) attributed to a misunderstanding in the interpretation and scoring of this indicator by the experts or (2) a possible limitation of the methodology. Perhaps it was a combination of both. The indicator "number of invasive species" would be assessed by the experts with an increase. Hence, the increase of invasive species reflects a decrease of the "pest and disease control". This discrepancy in terms of indicator and what is means for human wellbeing, was identified and address in the process of MESAT where some ES were "inverted" in terms of the indicators representing them (see Inácio et al. (2018) for details). In the Warnow estuary, the decrease is a product of an increase in the number of invasive species (from 7 to 11, Wittfoth 2011).

In the southern Baltic, while for some regulating and maintenance services (RM1, RM3 and RM7) a decreasing tendency can be observed, most services do not follow a pattern. It seems that each water body responds differently to the changes inflicted by anthropogenic and environmental conditions over the last decades. Eutrophication-related problems, the introduction of alien species by the shipping industry and other anthropogenic activities have led to the deterioration of some services (RM1, RM3 and RM7). Also, natural phenomena, such as mean sea-level rise and extreme rain events, have impacted the natural provision of other services (RM5). Despite the occurred ecological degradation, the last decades' efforts in enforcing EU environmental policies, such as the WFD and MSFD, have contributed to the improvement or maintenance of water bodies' ecological conditions and ecosystem services provision. Such example is the service RM6, which remained unchanged or increased due to the implementation of environmental protection networks such as the Natura 2000.

25.4.3 Cultural Ecosystem Services

Cultural services relate to the physical and spiritual use of the environment for human recreation, being in general associated with socio-economic, spiritual and historical activities. In the southern Baltic Sea, the 1960s' political situations were different from today. Until the reunification in 1990, Germany was divided into eastern (governed by the German Democratic Republic) and western (governed by the Federal Republic of Germany). Also, in 1990 Lithuania became independent from the Soviet Union and Poland established democratic governance. These political changes, together with increasing population density along the coast and increasing tourism numbers, led to an increase in cultural services provision in the southern Baltic Sea.

The increase in cultural services can be related to changes in human preferences. The service "experiential use of plants and animals" (Table 25.3, C1) is represented by the number of people taking part in in situ birdwatching activities. As of the time of the initial status, while visitors would travel to coastal lagoons to see big flocks of birds, there were no enterprises offering birdwatching tours and no statistical information. In the present time, birdwatching became a popular touristic attraction. For example, in the Curonian Lagoon, many tourists visit nowadays the cormorant colony in the Curonian Spit and the "Vėntes Ragas" ornithological station. In the Warnow estuary, there were fewer experiential activities related to nature, as in the 1960s the harbour and ship-building industry were in the development phase. Nowadays, NGOs and research-education centres offer activities related to biota around the estuary.

Another example is the service "physical use of seascapes in different environmental settings" (Table 25.3, C2). In the 1960s, the Szczecin Lagoon, Curonian Lagoon, Greifswald Bay and the coastal area of the Pomeranian Bay were either under the regimes of the GDR or the Soviet Union. In the 1960s in the GDR access to the coast was controlled during the day and prohibited during the night. After the reunification of Germany in 1990, Greifswald Bay, which during the GDR time was one of the few places in which water sports were allowed, gained more importance as a coastal tourism destination for water-related activities (StaluVP 2011; Fey et al.

Table 25.3 Assessment of ES provision changes for Cultural section at the class level for all MESAT case studies

	SE	WE	SL	GB	PB	CL
C1. Experiential use of plants (…)	5	3	0	3	5	3
C2. Physical use of seascapes (…)	4	5	4	5	3	2
C3. Scientific and educational	3	5	5	5	5	2
C4. Heritage, cultural	5	5	4	2	5	3
C5. Entertainment	5	5	2	5	0	1
C6. Aesthetic	5	5	5	5	3	5
C7. Symbolic	5	5	4	5	4	1
C8. Sacred and/or religious	0	-3	3	5	0	4
C9. Existence	0	0	0	0	5	3
C10. Bequest	5		5	5	5	4

SE Schlei Estuary, *WE* Warnow Estuary, *SL* Szczecin Lagoon, *GB* Greifswald Bay, *PB* Pomeranian Bay, *CL* Curonian Lagoon. Categories of change (CC) (based on the aggregation by reliability and for Curonian Lagoon aggregation by number): 0—no change; −1 to −5—decrease in services provision; 1 to 5—increase in services provision; grey colour—no data/not considered - adapted from Inácio et al. (2019)

2014). Nowadays the bay is an important hotspot for sailing and windsurfing, with around 20 sailing schools. In the 1960s, the Curonian Lagoon was part of the Soviet Union. After Lithuania became independent in 1990, the area around the lagoon developed and the Curonian spit was re-discovered as a tourism destination. Hence, the main economic activity in the area is connected to the port industry, therefore, there is not such a high increase in this service compared to other case studies (Table 25.3).

Another example is the service "Heritage, cultural" (Table 25.3, C4). Its increase is related to the increasing number of culturally important sites. With lifestyle shifts, some traditions and cultural activities have been abandoned through time. To keep these alive, they became important to preserve as heritage. Political and socio-economic changes are also responsible for the easier access and education and development of science. This is represented in the service "scientific and educational" (Table 25.3, C3). In all case studies, the number of scientific publications related to the water bodies has increased significantly compared to 1960s.

Technological advances can also influence changes in cultural services. The service "Aesthetic" (Table 25.3, C5), for example, increased from the initial to present status. This service is represented by the number of pictures related to the water bodies. In the initial time, not everyone had easy access to a photo camera. Therefore, the image representations of the water bodies were either from paintings or postcards and not necessarily about nature (Schernewski et al. 2019). Nowadays, the easy access to photo cameras increases the number of pictures related to the case studies, which can be accessed via geotagging. Therefore, this indicator represents more of a technological advance rather than a change in provision. We decided to keep this indicator, as a compromise between feasibility and data availability. Nevertheless, it still provides information and an interesting perspective on the representation and "access" of the same ecosystem service between two time periods.

In the southern Baltic Sea, important socio-economic and political changes led to an increase in provision of cultural services. However, it does not mean that the capacity of the water bodies to provide these services has increased. It rather indicates increasing human access and demand for coastal cultural outputs (Garcia Rodrigues et al. 2017). However, this increased demand may affect the provision of other services (regulating and maintenance and provisioning). Assessing temporal changes in cultural ecosystem services can unveil the value and importance, in this case of southern Baltic water bodies, for people's wellbeing. This may be of importance when mainstreaming the necessity of achieving environmental targets.

25.5 Lessons Learnt: The Role of Time in Ecosystem Services Assessments

What can the past tell us? Comparing a habitat or water body at two or more points in time, using ecosystem services assessments provides a deeper understanding of system structure, function and behaviour. Dynamics, trends and as well as trade-offs between services become visible and, potentially, can be associated with socio-economic drivers.

Which were the drivers of change? In the southern Baltic region, the developments during the last 150 years were controlled by events and long-term processes. Major events were World War I and II and, more recently, the political changes. During the late 1980s, Germany became reunited and Poland and the Baltic States changed from socialistic countries into democratic states and market economies. However, these events only caused disruptions and overlaid ongoing long-term developments: the population in the southern Baltic and the concentration in the coastal zone strongly increased during the last century. Urbanized and industrialized coastal areas spread and reduced natural habitats. Food demand increased, causing expansion and intensification of fisheries and especially agriculture. Consequences are ongoing pollution, eutrophication and destruction of habitats in the sea. In coastal areas during the last 50 years, the industrial is transforming into a leisure society. Tourism is further gaining importance, nature aesthetics became more important and the environmental awareness increased. Nature preservation and sustainable development moved to the foreground. These developments become visible in our ecosystem service assessments, but the pattern differs between different regions, habitats or waterbodies.

Challenges of historical assessments? Despite many benefits, there are very few studies of historical assessments of ecosystem services. If assessing coastal and marine ecosystem services for the present time is already challenging, it becomes harder for the past. Based on the lessons learned, the availability and inconsistency of data in time and space is perhaps the biggest challenge to overcome. Another challenge has to do with meaning and suitability of indicators, which may change over time and differ among water bodies. Nevertheless, a major challenge is the definition of a reliable historic baseline for assessing changes in ecosystem services provision.

Do reliable baselines in time exist? In ecology, the concept of baselines as reference for assessments is common. In the WFD, reference conditions describe the biological quality elements in coastal water types at high ecological status. The objective of defining reference conditions is to enable the assessment of ecological quality against these standards (CIS 2003). The reference conditions serve as a basis for the definition of the good ecological status, which is the target of restoration measures. Already at an early stage of the WFD implementation, it became obvious, that no coastal waters exist that could serve as a reference. As a consequence, historical ecological reference conditions had to be defined and a concrete point in time had to be determined as reference. In the southern Baltic waters, it corresponds to the years around 1880. Today, this baseline is scientifically and societal well accepted.

Can 1880 serve as a baseline for ecosystem service assessments? The description of the ecological status of coastal waters in the years around 1880 within the WFD was a complex process (Schernewski et al. 2015). It required a combination of modelling, extrapolated historical data and expert judgement. Ecosystem services cover a much broader thematic spectrum and the ecological state is only one aspect. Ecosystem service assessments require a broad range of data which hardy exists for the year 1880. The further back in time the baseline is the less reliable is the existing knowledge and subsequent assessment, or it remains incomplete. Another aspect is that several indicators describing an ecosystem service are not applicable for this period. This is especially true for cultural service indicators that simply did not exist yet, such as movies and broadcasts, pictures, Red List and iconic species or marine protected areas. Therefore, we do not consider 1880 to be a suitable assessment baseline.

Is the good ecological status of water bodies a suitable assessment baseline? Schernewski et al. (2015) provide information about the nutrient and single biological element values defining a good ecological status. Comparing these values with historical data indicates that in the outer coastal waters and in some lagoons and estuaries a good ecological status still existed in 1960. However, several water bodies, such as the Schlei were already heavily eutrophied. Other water bodies such as the Kieler Förde or the Warnow estuary were already heavily industrialized in 1960. As a consequence, the 1960s cannot be used as a universal baseline reflecting a good ecological status of southern Baltic coastal waters. Despite that, the 1960s are well suitable as a baseline in time, or initial status, for ecosystem service assessments. Data availability is already sufficient, and the distance in time to the present situation of 60 years is long enough to reflect changes in ecosystem service provision.

The WFD as the backbone for ecosystem service assessments? The WFD not only provides temporal baselines, a complete spatial subdivision of the seascape, but also large amounts of data, maps and information that were gathered and compiled during its implementation process. It is implemented across the entire EU, which allows a transfer of the approach to other countries and coastal waters. This altogether forms an outstanding basis for ecosystem service assessments in coastal and marine waters. Last but not least, the WFD defines concrete end-users for assessment results and ensures a practical relevance of results. Similar to the WFD, the Biodiversity Strategy 2020 has the aim to restore ecosystems and their services, by achieving a sustainable and desirable ecological state. It initiated a mapping of ecosystem services all over Europe. Adding a temporal, historic perspective to this spatial mapping has obvious benefits because it allows the elaboration of biodiversity restoration targets and gives an insight into the potential of a system to provide services.

MESAT—a step forward? MESAT is designed for coastal and marine water systems and has, concerning the choice of services, indicators or reliability, all general strengths and weaknesses associated with ecosystem service assessments. The relative comparison of two ecosystem states allows single experts to carry out fast assessments by integrating different types of data and information. It includes a supporting visualization, a stepwise aggregation and a reliability assessment. It is flexible, easy to apply and can be used beyond the scope of this article, e.g., for comparing different management scenarios, future states or contrasting aquatic systems (Inácio et al. 2018, 2019; Robbe et al. 2018; Schernewski et al. 2018, 2019).

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Assessing Temporal Changes in Ecosystem Service Provisions: Conceiving Future Pathways

26

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Abstract

Due to the close interrelation between humans and the environment, social-ecological issues are of great importance for environmental science and management. In this context, anthropogenically induced environmental pressures are omnipresent. Examples are conventional agricultural production, with consequences such as loss of biodiversity, erosion, eutrophication, as well as the burning of fossil fuels, which causes anthropogenic climate change. Moreover, in this context, coastal ecosystems—and as such our Baltic coast—are particularly vulnerable. In the following section, some insights into the vulnerability of coastlines notably concerning the contemporary drivers of climate change and intensive agricultural production are presented. Furthermore, this

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chapter will provide a glimpse into our potential future focusing on estimating future capacities of coastal ecosystems to provide ecosystem services through scenario assessments. More precisely, (i) the influences of the scenario conditions on the ecosystem service potentials are estimated, (ii) the potentials under the different future pathways are compared to each other and (iii) the sensitivity of the assessed ecosystem types to the set of scenario conditions is assessed.

26.1 Our Coastal Zone: Influenced by Climate Change and Agricultural Production

Coastal zones, functioning as the connection between the hinterland and the open ocean, form unique environments (Jurasinski et al. 2018; Chaps. 3 and 4 of this volume). As they are influenced by terrestrial as well as marine processes, they are very dynamic and strongly exposed to pressures and threats such as eutrophication and habitat degradation (HELCOM 2018a, b; Snickars et al. 2014). In the following, some insights into the influence of climate change and intensive agricultural production on the German Baltic coastal zone are outlined.

26.1.1 Climate Change

Next to sea level rise and changes in air and sea surface temperatures, climate change will lead to further alterations in the German Baltic Sea area. Climate change affects the quantity and spatio-temporal patterns of precipitation (Sein et al. 2014; RADOST-Verbund 2014; DWD 2017, 2018). Up until now, primarily winter precipitation has increased in the German Baltic Sea area (HZG 2012). In particular, heavy precipitation events lead to increased erosion and runoff into coastal systems. Most likely seawater salinity and pH are continuing to decline (Sein et al. 2014; HELCOM 2018b). As a consequence of the changes in temperature and pH, biodiversity is affected, and the food web is most likely altered (HELCOM 2013; Snickars et al. 2014; Bauer et al. 2019). Besides, through global warming, low oxygen levels near the Baltic seabed might be amplified, intensifying eutrophication effects (HELCOM 2018a). Furthermore, due to sea-level rise, the frequency of storm surges and the intensity of cliff erosion are likely going to increase (Łabuz 2015; HZG 2012; Hoffmann and Lampe 2007). Further implications of climate change on the German Baltic coastal waters are increasing risks for bathing water quality through microorganisms and possibly increasing potentially human-pathogenic vibrio bacteria and/or new human pathogens, which benefit from higher temperatures (Sterk et al. 2015; Vezzulli et al. 2016).

26.1.2 Agricultural Production

Conventional agricultural production is featured by monocultures, intensive tillage practices and livestock farming, high fertilization rates and the application of pesticides. All of these aspects can be considered as pressures on our ecosystems (Ericksen 2008; German Environment Agency 2015; Taube 2016; Turner et al. 2016; Therond et al. 2017; Augstburger et al. 2018; Böhning-Gaese et al. 2019). The application of plant protectants, e.g. in the form of pesticides, endangers biodiversity and surface water quality (FAO 1996; Power 2010). Through the vast application of nutrients onto agricultural grounds, short-term biomass production can be increased (Vitousek et al. 2002). Nevertheless, through nutrient oversupply in agricultural systems, high nutrient losses are generated. Nutrient oversupply poses serious threats to the environment as nitrogen and phosphate degrade inland- and coastal water quality and endangers biodiversity (Sutton et al. 2013; BLANO 2014; Taube 2018; De Notaris et al. 2018; Friedland et al. 2019). The enrichment of nutrients in water bodies leads to eutrophication (Welte and Timmermann 1985; Chislock et al. 2013; Dominati 2013; Jónsson and Davídsdóttir 2016; HELCOM 2018a). Eutrophication of our coastal waters leads amongst others to increased water turbidity and primary production of phytoplankton as well as oxygen deficiency (Snickars et al. 2014; BLANO 2014; HELCOM 2018a, b; Friedland et al. 2019). As the Baltic Sea is a relatively closed system (Meier et al. 2019), which is only connected to the North Sea via the Skagerrak/Kattegat Strait, nutrient inputs remain virtually trapped in the system due to the long residence times of the water (BLANO 2014; HELCOM 2018a, b).

Summing up, both climate change and agricultural production can be considered as highly relevant driving forces shaping the German Baltic Sea area (Snickars et al. 2014; Latacz-Lohmann et al. 2019; Chap. 3 of this volume). Thereby, they influence future capacities of the affected ecosystems to self-organize and provide ecosystem services. Next to scientific objectives, the generation of knowledge on the effects of both drivers on the systems is also extremely relevant in the socio-political context. Fundamental knowledge and deep expertise on the human-environmental system are required to develop strategies aiming for a sustainable future of the German Baltic Sea catchment area (European Commission 2016; Costanza et al. 2017; Chap. 5 of this volume).

26.2 Ecosystem Service Scenario Assessment

The ecosystem service concept has been proven to be a suitable approach for assessments with regard to both climate change and agricultural pressures (Tucker et al. 2009; Power 2010; Martinez-Harms et al. 2017; Willemen et al. 2017; Bicking et al. 2018, 2019, 2020). Thus, the concept sets an expedient framework for the assessment of the influences of the drivers of intensive agricultural production and climate change in the German Baltic Sea area. In combination with scenarios, assessments can provide useful means of translating alternative future pathways of drivers into projected consequences for our environment and ecosystem services (IPBES 2016).

Ecosystem services can be differentiated into attributes of potential, flow and demand. Whereby, the potential refers to the hypothetical maximum yield of the individual service (Burkhard et al. 2014). The flow, which is driven by a certain demand for the ecosystem service, defines the actually used or harvested service (Syrbe et al. 2017). Within this chapter, the assessment focusses on analyzing the influences of the considered drivers on the ecosystem service potentials. Basing on the investigations of Schumacher et al. (Chap. 24 of this volume), for 2050 and 2100, changes in the potential of ecosystems to provide services are analyzed based on six exploratory scenarios. These scenarios are used to assess a range of plausible futures considering climatic changes as well as adaptations in political framework conditions regarding climate protection and agricultural production.

Generally, the assessment is based upon the German Baltic Ecosystem Service Potential Matrix (Baltic ESP Matrix), introduced in Chap. 24, which links ecosystem service potentials to specific terrestrial land use/land cover classes and coastal and marine ecosystems (Chap. 24 of this volume; Müller et al. 2020; Burkhard et al. 2014). The qualitative scenario assessment does not account for land use/land cover change nor does it account for spatial differentiations within the same land use/land cover class or rather ecosystem type. These limitations imply that the scenario assessment is no complex high-resolution modeling approach, but rather a **thinking experiment** trying to grasp a glimpse into potential futures.

Recent studies and assessments in the Baltic Sea area relied on high resolution and complex modeling techniques (a.o. Chaps. 3, 4 and 6 of this volume; The BACC II Author Team 2015; Allin et al. 2017; Meier et al. 2019). In this context, studies assessed future climatic conditions and the ecological state of the study area (Chaps. 3 and 4 of this volume; Allin et al. 2017; Friedland et al. 2019). These kinds of assessments deliver high quality and very specific information on the observed variables. Nevertheless, approaches like these are very data- as well as time-consuming and require extensive technical skills. As various stakeholders and user groups do not possess these required technical and professional skills, the need for a more user-friendly and accessible method, such as the Baltic ESP Matrix, arose (Chap. 24 of this volume). Of course, such a simplified and qualitative approach will never replace the comprehensive and precise model-based assessments. It rather complements the picture and enables users a first rudimentary evaluation of the study area, in particular through the simplicity and transparency of the approach (Müller et al. 2020). Therefore, the results of this assessment should be understood as relative tendencies comparing the different scenarios rather than absolute changes and/or site-specific forecasts. In addition to that, the assessment is not based upon the most probable and realistic scenario conditions, which are typically used in the regional studies involving scenario assessments, but on rather extreme (to the low- and high-end) anthropogenic radiative forcing conditions (DWD 2020). Therefore, this

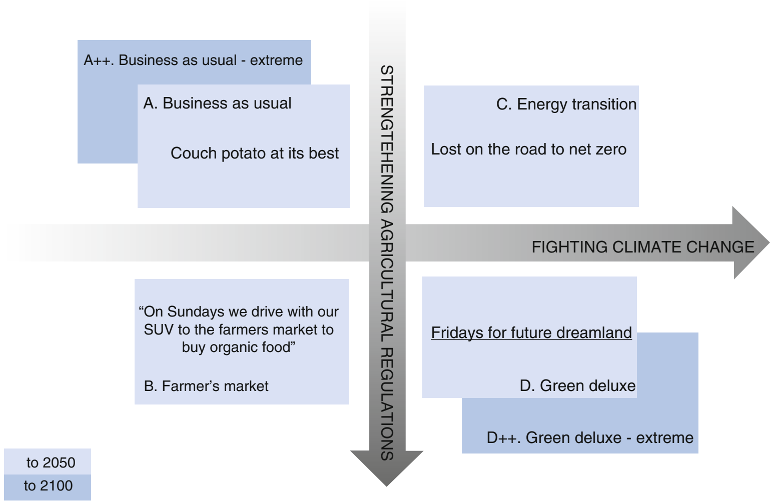

Fig. 26.1 Schematic overview of the developed scenarios

assessment does not aim to project our most realistic future but rather highlights the potential variances of the great variety of future pathways. Despite these limitations and the strong respective uncertainties, the scenario assessment can contribute to problem identification as well as agenda-setting.

26.3 The Scenarios

The development of the scenarios has primarily been based upon the climate reports for Schleswig-Holstein and Mecklenburg-Vorpommern (DWD 2017, 2018), the results of the RADOST[1] project (RADOST-Verbund 2014) and the reports published by HELCOM[2] (HELCOM 2018a, b) and BLANO[3] (BLANO 2014).

In total six different scenarios have been developed. In Fig. 26.1, they are plotted based upon realized measures with regard to climate change and agricultural production. The scenarios A, B, C and D have been constructed for the year 2050, whereas the scenarios A++ and D++ refer to the year 2100.

Paraphrasing the scenarios, both A and A++ depict a *business as usual* situation. Thus, the societal processes are to a large extent still based upon the combustion of fossil fuels and organic farming is only performed on approximately 6–9% of the agricultural area. Today's agricultural regulations concerning fertilization are still in place. Nevertheless, the climatic conditions vary between A and A++, as these

[1] Regional Adaptation Strategies for the German Baltic Sea Coast

[2] Baltic Marine Environment Protection Commission

[3] Federal/Länder Committee on the North Sea and Baltic Sea

Table 26.1 Overview parameters considered in the scenario assessment

	A	A + +	B	C	D	D + +
Reference year	2050	2100	2050	2050	2050	2100
Air temperature	++	+++	++	+	+	+
Sea surface temperature	++	+++	++	++	++	++
Precipitation	++	+++	++	+	+	+
Nutrient surpluses	+/−	+/−	−	+/−	−	−
Organic agriculture (incl. Biodiversity conservation)	+/−	+/−	+	+/−	+	++

+++, ++ and + represent high, medium and low increase, − - -, − - and - represent high, medium and low decrease and +/− represents no significant change

follow the Representative Concentration Pathways (RCP, IPCC 2018) for the specific reference years.

In contrary to the A scenarios, in scenario B, *Farmer's market*, additional regulations are implemented aiming to increase the multifunctionality of agricultural landscapes, which involve measures concerning fertilization practices, organic farming and biodiversity conservation. Examples are production-integrated measures such as flower strips, nature conservation on the landscape scale, as well as general changes in the agricultural production system supporting resource efficiency. Nevertheless, no additional measures are taken to combat climate change. Scenario C, *Energy transition*, on the other hand, depicts a future without these additional agricultural regulations in place, but with measures implemented to fight climate change. In the *Green deluxe* scenarios (D and D++), strong agricultural regulations, as well as measures against climate change, are implemented. Again, they follow the respective RCP concerning the reference years 2050 and 2100.

To be consistent with the official climate reports of the two federal states with regard to the atmospheric greenhouse gas concentration pathways, the RCPs 8.5 and the 2.6 have been considered (DWD 2017, 2018). The scenarios A, A++ and B follow the RCP8.5, whereas the scenarios C, D and D++ follow the RCP2.6. In Table 26.1, the changes of the considered parameters such as temperature and nutrient surpluses under the scenarios are presented. For a matter of simplicity, no regional differences have been considered, but the same scenario conditions apply for the whole study area. In order to illustrate the development of the scenario conditions, the projected changes for precipitation are exemplarily outlined. According to the DWD projections, annual mean precipitation in the study area increases by around 2% and 5% until 2050 and by 2% and 10% until 2100, under the RCP2.6 and RCP8.5, respectively (DWD 2017, 2018). Translating these projected changes into our scenario conditions, we obtain the ranked relative changes "+", "++" and "+++" (see Table 26.1), corresponding to low, medium and high increased annual mean precipitation.

Within the assessment, the projected changes concerning the climatic conditions and agricultural production are translated into changes in ecosystem service potentials. Thereby, only the relative changes noted in Table 26.1 are considered.

As outlined in Chap. 24, the original evaluation is based upon a normalized scale ranging from 5 (no relevant potential) to 90 (very high relevant potential). For this scenario assessment, the original potential values of the ecosystem services for the individual ecosystem types have been modified. The modification is based upon expert evaluation by an interdisciplinary group of scientists, which has been carried out in the form of roundtable discussions. The group of experts consisted out of seven experts from different research domains, e.g. (physical) geography, coastal and marine research, agricultural sciences, and biology. The climatic and agricultural scenario conditions outlined in Table 26.1 have been weighted evenly for the evaluation. In order to provide a frame suitable for the normalized scale from 5 to 90 (Chap. 24 of this volume), the evaluation is performed in terms of addition or subtraction of potential values between zero and twenty, considering the individual scenario conditions. The changes are restricted to these values in order to prevent the assessed ecosystem to lose their ecological and physical characteristics as well as to prevent non-scientific exaggerations. The changes have been adjusted according to the general census of the roundtable, whereby the evaluation was executed in several rounds and the results were reviewed repeatedly. As an example, under the scenario conditions D++, which are only slightly increasing temperature and precipitation, decreased nitrogen surpluses and increased organic agriculture, the potential of the ecosystem service natural heritage is thought to increase on arable land compared to the reference state.

26.4 A Glimpse into the Future

The assessment reveals diverging projections concerning future ecosystem service potentials (Fig. 26.2). Generally, the potentials of the individual ecosystem services strongly diverge within the nine aggregated ecosystem types. Lowest sets of potentials can be found in settlements. The provisioning ecosystem services fish and seafood, as well as crop production, only deliver high potentials in specific ecosystem types, which are agroecosystems and inland as well as coastal waters, respectively. The distribution of potentials throughout the different ecosystem types is less distinct for the other ecosystem services. Nevertheless, in particular natural ecosystems are featured by high ecosystem service potentials.

Under each scenario, a unique set of ecosystem service bundles is provided (Fig. 26.2). Generally, the A and A++ scenarios deliver the lowest ecosystem service potentials. An exception to the rule is the ecosystem service crop production, which peaks in agroecosystems under the scenario A. Nevertheless, fast-forwarding the *Business as usual* scenario in time up to 2100 (A++), the potential for the provision of all other ecosystem services throughout the different ecosystems drops dramatically. This is also true for the provision of fish and seafood. Under the scenario conditions B and C, the ecosystem services potentials are mostly at a mediocre level.

Largest differences of ecosystem service potentials between these two scenarios can be found in the agroecosystems, wetlands and inland as well as coastal waters. Next to the ecosystem service global climate regulation, the potentials deviate most

Fig. 26.2 Potentials of selected ecosystem services under the different scenarios for aggregated ecosystem types (the longer the bar, the greater the potential)

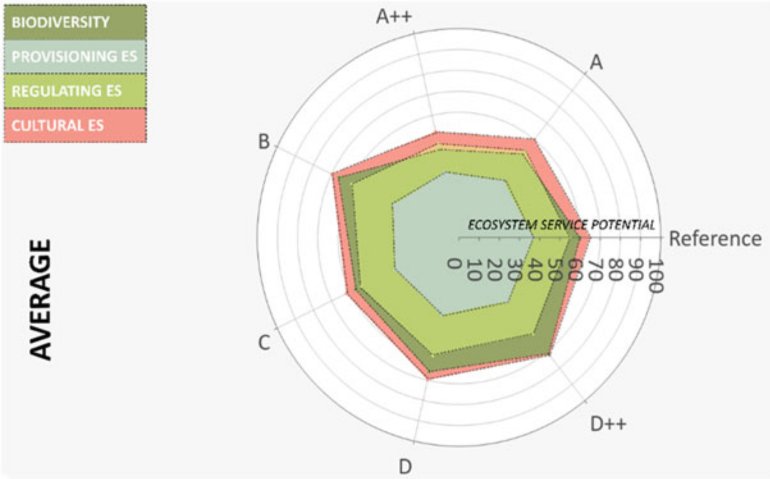

Fig. 26.3 Comparison of averaged potentials of biodiversity, provisioning, regulating and cultural ecosystem services under the different scenarios

between scenarios B and C, with respect to nutrient regulation, natural heritage, crop production and biodiversity. Overall, comparing the scenarios B and C a somewhat more balanced bundle of ecosystem service potentials can be found under scenario B, *Farmer's market* (Fig. 26.3).

Concerning the cultural ecosystem services (landscape aesthetics & inspiration and natural heritage), in particular the implementation of strict agricultural measures seems to result in increased potentials. In some cases, the D and/or D++ scenarios even outperform the reference state. Across the board, the greatest differences in ecosystem service potentials can be found when comparing the two scenarios with the reference year 2100 (Figs. 26.2, 26.3 and 26.4).

Under the D++ scenario conditions, individual ecosystem service potentials are significantly higher than under the A++ scenario circumstances. This trend can be seen in both terrestrial and coastal/marine environments. On average, the ecosystem service potentials of the aggregated ecosystem types are 26% lower under A++ scenarios conditions compared to the potentials under the D++ scenario conditions. Comparing the A++ and D++ scenarios for landscape aesthetics & inspiration, the ecosystem service potentials differ most in agroecosystems and inner as well as coastal waters (see Figs. 26.2 and 26.4).

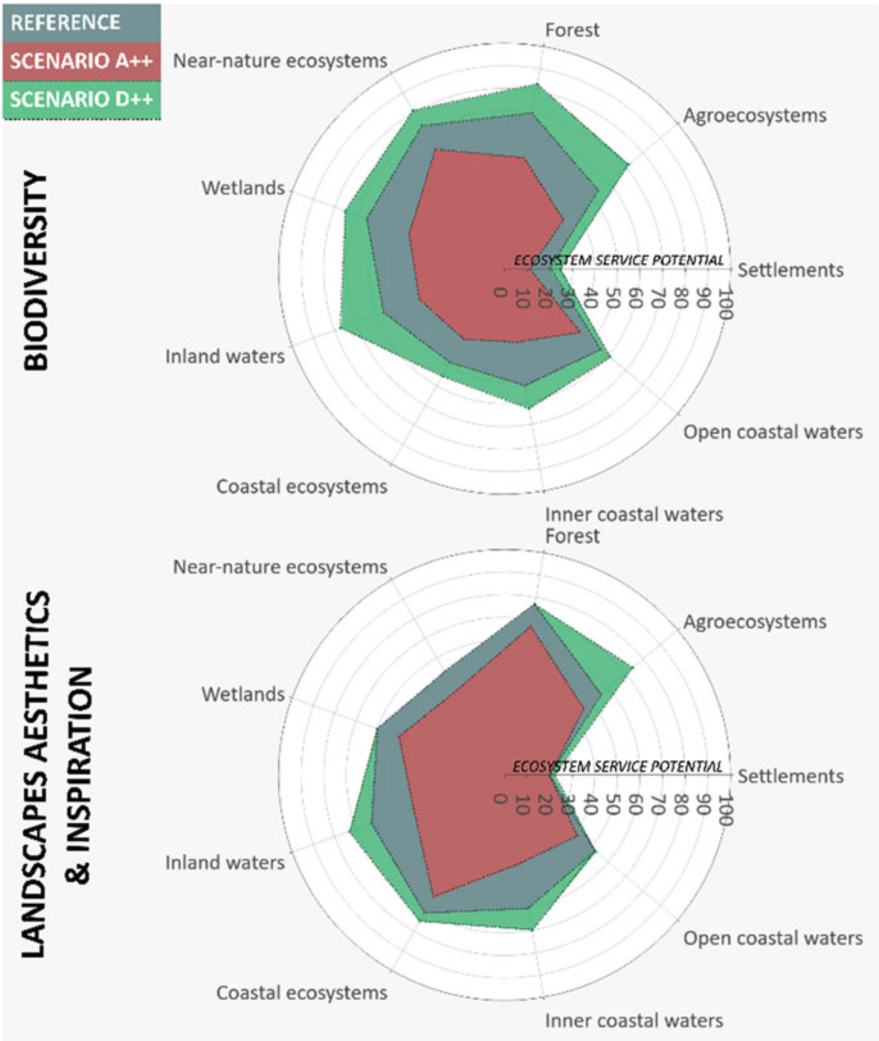

Fig. 26.4 Comparison of potential profiles for biodiversity and the ecosystem service landscape aesthetics and inspiration under the scenario conditions A++ and D++

Box 26.1 Caution—Hot!

We would like to emphasize once more that this assessment has been performed as a "thinking experiment", comparing extreme future pathways based upon expert evaluation. The changes of the ecosystem service potentials are solely based upon the estimations of the experts considering the direct

(continued)

Box 26.1 (continued)
consequences of the projected scenario conditions presented in Table 26.1. Considering both, the high level of abstraction and the very specific scenario conditions, it is not surprising, that the results of the assessment partly do not agree with recent regional scenarios and projections aiming to project highly probable conditions and changes (e.g. The BACC II Author Team 2015; Allin et al. 2017; Meier et al. 2019).

To assess the variability of the ecosystem service potentials under the different scenarios, the medium standard deviation of the projected potentials has been calculated for the ecosystem types and individual ecosystem services (see Fig. 26.5). With reference to the assessed ecosystem services in particular natural heritage, nutrient regulation and biodiversity feature high standard deviations. Concerning the aggregated ecosystem types, the highest standard deviation can be found for agroecosystems (Fig. 26.5). This finding was to be expected as a large share of the scenario conditions directly apply to the management of these systems. The next highest standard deviations can be found for inland and inner coastal waters (Fig. 26.5), indicating their sensitivity to the assessed future pathways.

The scenario assessment also allows for spatial analysis of the changing ecosystem service potentials under the diverse scenario conditions. In Fig. 26.6, the spatial distribution of the average standard deviation of the projected changes for the assessed ecosystem services is presented. As the agroecosystem types non-irrigated arable land and pastures are the most dominant land use/land cover

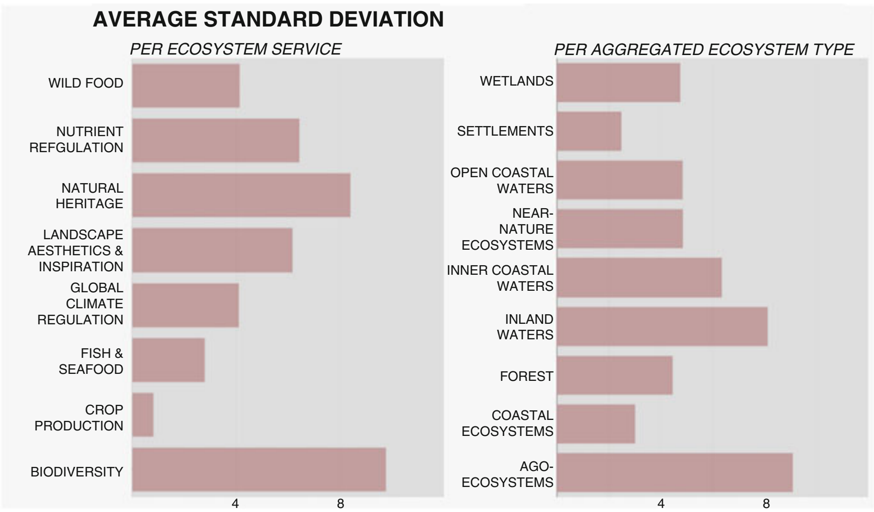

Fig. 26.5 Average standard deviations of projected ecosystem service potentials under the scenarios (left) per ecosystem service and (right) per aggregated ecosystem type

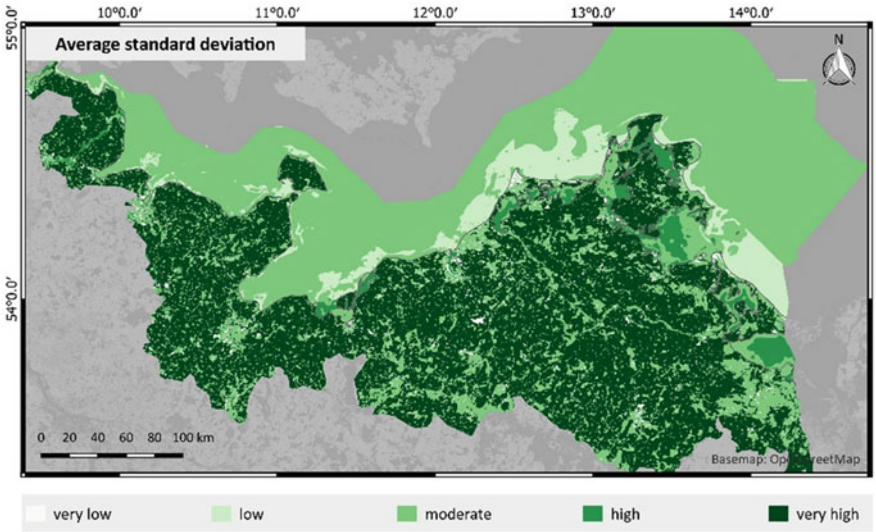

Fig. 26.6 Average standard deviation of assessed ecosystem service potentials (incl. biodiversity) across all six scenarios

classes in the terrestrial part of the study area, the respective area is to the largest extent characterized by very high standard deviations. Only small fragmented patches are characterized by very low to moderate standard deviations. In particular, the coastal area in Mecklenburg-Vorpommern delivers the most heterogeneous distribution, ranging mainly between moderate to high standard deviations. The greatest share of the marine part of the study area ranges from low to moderate standard deviations.

In the following box, the results of a case study in the German Baltic Sea area with respect to the ecosystem service global climate regulation are presented.

Box 26.2: Blue carbon potential in the German Baltic Sea zone: a case study

Case study coordinator: Ana Belén Almagro.

Carbon sequestration and storage in the context of coastal systems is referred to as blue carbon (Nellemann et al. 2009; Burkhard and Maes 2017). Lately, the relevance of blue carbon in the fight against climate change has been highlighted (Stigson et al. 2015) as recent studies have shown that

(continued)

Box 26.2 (continued)

coastal ecosystems might have the potential to sequestrate more carbon per unit area than terrestrial forests (Mcleod et al. 2011). In this case study, we considered the projected conditions up to 2100 under the two scenarios, A++ and D++. In a first step, the Sea Level Affecting Marshes Model (SLAMM 6.7, Warren Pinnacle Consulting Inc. 2016) was used to map changes in the spatial distribution of coastal ecosystem types under SLR projections considering dry land defense. In a second step, these projections have been used for the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST 3.8.0, Sharp et al. 2020) Coastal Blue Carbon model (CBC). Within the InVEST CBC model, the projected habitat distribution, tabular data from an extensive literature review on e.g. carbon pools, transitions of habitats, and transient of the carbon pools are processed and combined to provide information about carbon stocks, emissions, and sequestration for the defined assessment period.

By the year 2100, habitats across the German Baltic zone are predicted to be inundated or eroded to other categories on different levels depending on the considered SLR scenario (Fig. 26.7). The A++ scenario, considering a SLR of 0.98 meters, is featured with changes in some habitats such as transitional salt marshes and regularly flooded marshes that are of considerable relevance to blue carbon sequestration and emission rates (Mitra and Zaman 2014). When ecosystems are converted or degraded, they release stored carbon into the atmosphere, thereby turning into a source of greenhouse gases (IPCC 2019). Under the D++ scenario conditions, a SLR of only 0.26 meters is considered (DWD 2017). Consequently, the predicted variations are not as drastically diverse from the initial condition. Under the different future pathways, the projected ecosystem service potentials concerning blue carbon vary a lot. The projected net carbon sequestration between the two specified years is presented in Fig. 26.8. The A++ *Business as usual—extreme* scenario displays more unfavorable conditions for the mitigation of climate change. More precisely, under the A++ scenario conditions, the net carbon sequestration between 2019 and 2100 in non-dryland ecosystems (as of the habitat distribution in 2019) is around 10% lower compared to the net carbon sequestration under the D++ scenario conditions. Thus, the ecosystems lose a great deal of their inherent carbon sequestration and storage capacity. These results highlight the relevance of the successful implementation of sustainable strategies fighting climate change (IPCC 2019).

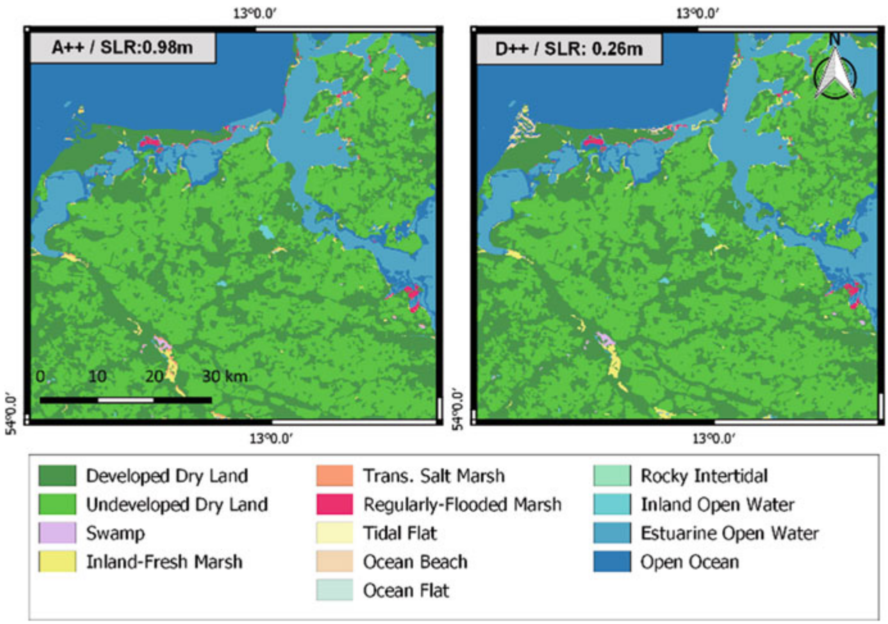

Fig. 26.7 Excerpt of the projected habitat distribution under the two scenarios A++ Business as usual—extreme (left) and D++ Green deluxe—extreme (right) for the year 2100

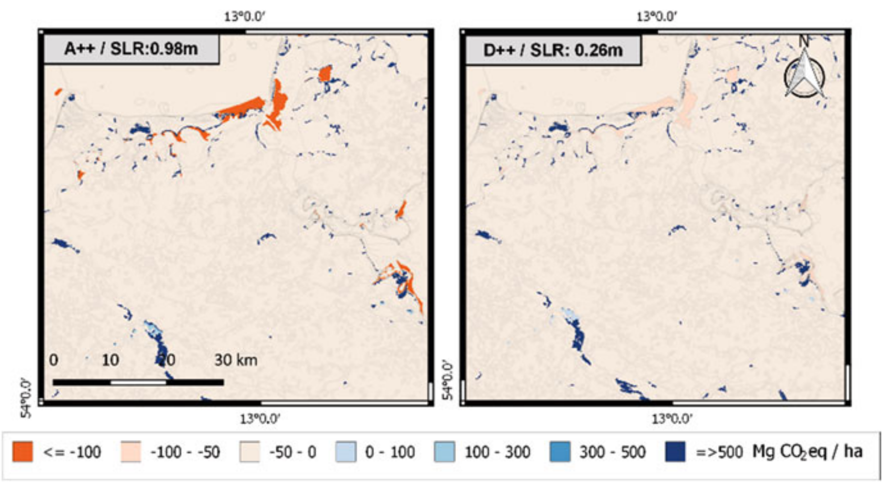

Fig. 26.8 Excerpt of the net carbon sequestration between 2019 and 2100 under the two scenarios A++ Business as usual—extreme (left) and D++ Green deluxe—extreme (right)

26.5 Our Future Is Not Fixed

This assessment presented an overview of the provoked pressures of the two contemporary drivers: climate change and intensive agricultural production onto the German Baltic Sea area. In that context, six different scenarios with respect to climate change and agricultural production have been set up to present potential changes. These changes involve amongst others nutrient surpluses and climatic variables such as temperature and precipitation. Based on these scenarios, (i) the influences of the scenario conditions on the ecosystem service potentials were estimated, (ii) the potentials under the different future pathways were compared to each other and (iii) the sensitivity of the assessed ecosystem types with respect to the different scenarios were assessed. Under the different scenario conditions, unique sets of ecosystem service bundles are provided. Cultural ecosystem services (landscape aesthetics & inspiration and natural heritage) seem to be most strongly influenced by the agricultural scenario conditions. In particular the long-term scenarios (until 2100) *Business as usual—extreme* (A++) and *Green deluxe—extreme* (D++) delivered significantly diverging results. Under the green utopian scenario (D++), the assessment delivered the highest ecosystem service potentials, whereas the dystopian scenarios (A++) delivered for the most part reduced potentials, also compared to the reference state. In addition to that, the assessment revealed, that the variability of the projected ecosystem service potentials under the diverse scenarios is next to agroecosystems, highest in inland and inner coastal waters. This aspect highlights the vulnerability of these systems concerning future decision-making and land management strategies.

The developed scenarios did not follow the most probable and realistic future projections but have been set up based upon rather extreme scenarios. Concerning the climatic conditions, the scenarios followed the RCPs 2.6 and 8.5 and considering agricultural production, nowadays conventional agricultural production was compared to a utopian organic agricultural system that aims at producing ecosystem service bundles rather than aiming to optimize crop production. In addition to that, the assessment is based on literature research and expert evaluation. As outlined in Chap. 24, expert evaluation is accompanied by a high degree of uncertainties and subjectivity. The evaluation might be biased based on personal normative loadings and their knowledge backgrounds. We have tried to limit the uncertainty of the expert evaluation and to capture as comprehensive as possible the general system understanding through consulting experts from different research domains. Nevertheless, the number of experts was still very limited and contributed considerably to the uncertainties of this study. The evaluation format, roundtable discussions, supports knowledge-exchange and facilitates the incorporation of an interdisciplinary perspective in the process. Nevertheless, the format, unfortunately, did not allow for a precise detection of the level of uncertainty of the evaluation.

In addition to that, the assessment could not account for the complexity of the whole system, such as the numerous (in-)direct interrelations and aspects of seasonality. Thus, the results of the assessment need to be handled with care. The scenario assessment should give us a feeling and understanding of the variability of our future pathways and should highlight the relevance of expedient planning and decision-making considering the production of ecosystem service bundles. The insights provided by this chapter highlight, in particular, the relevance of climate mitigation measures, sustainable land management policies and agricultural practices. They are key to prevent environmental pollution, to prevent the degradation of soils as well as inland and coastal water bodies and to sustain the capacity of our ecosystems to provide ecosystem services.

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Part V

Synthesis: Assessment as a Tool for Managing Coastal Ecosystems?

27 Applying the Integrated Approach

Hendrik Schubert and Felix Müller

Abstract

This chapter will introduce the last part of this volume, closing the full circle of papers by critically evaluating the outcomes of the combined efforts of a comprehensive analysis of ecosystem services provided by the coastal systems of the Southern Baltic Sea. After introducing the reader into the social-ecological aspects of coastal ecosystem uses in Chap. 2, an in-depth presentation of the most recent results with regard to their ecology were given in the Chapters of Part III. Together with the economical and ethical aspects dealt with in the Chaps. 19–23, this became the basis of an ecosystem service assessment presented in detail in the Chaps. 24–26.

Intended as a tool for balanced decisions with regard to sustainable spatial planning within coastal zone management (Schernewski and Schiewer 2002, Schernewski et al. 2011), ecosystem service assessments are supposed to include most aspects of societal demands (Burkhard et al. 2012, Burkhard and Maes 2017). Consequently, the outcome of such an assessment not only broadens the horizon of disciplinary stakeholders, it often also raises concerns about respecting their individual interests.

If not being treated carefully, such concerns and processes may result in lack of acceptance of the outcome. On the other hand, lobbying by a well-organized group of stakeholders may result in overrepresentation of their specific demands, leaving

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H. Schubert, F. Müller (eds.), *Southern Baltic Coastal Systems Analysis*, Ecological Studies 246, https://doi.org/10.1007/978-3-031-13682-5_27

the others with a feeling of being treated unfair. A kind of worst-case scenario comes into play when scientific results contradict expectations. In such a case, the initial hypotheses should be rejected. However, that is not always the case, sometimes criteria are re-defined until the results fit the prejudices (a classical example for this is presented by Gould and Lewontin (1979). But there are also many examples how to treat such problems the correct way. Best known is probably the biodiversity debate where it has been shown that human activity *per se* does not have to lead to a decrease of alpha-diversity locally and regionally. In fact, urban areas can be diversity-hotspots as shown by, e.g. Schubert (2013). Dune ecosystems in touristic areas can exceed diversity of undisturbed ones (Grunewald 2004; Grunewald and Łabuz 2004). Rejecting the initial hypothesis in these examples leads to an analysis of reasons as well as consequences and the gained knowledge resulted in a more detailed view of "biodiversity" by species-specific valuation (Grunewald and Schubert 2007; Genovesi and Shine 2003; Weber et al. 2005; Nehring 2016).

The above examples are caused by lack of knowledge, solved by further investigations. Such fields of lack of knowledge became obvious in Chap. 6, when experts initially largely differed in their assessment of individual ecosystem services. Some of them could be solved by raising awareness for details not commonly known, others are principal problems, which will be treated here.

One main principal problem not yet completely solved for all kinds of ecosystems are the mechanisms ecosystem services are produced with. Especially the link between ecosystem functions and ecological integrity on the one side and ecosystem service provision on the other is of high interest and studied intensively in order to overcome the expert opinion-based assessments, which restricted ES assessments to a more descriptive tool with low forecasting potential. This problem will be tackled in the following Chap. 28 in detail, using the community structure of coastal systems as focal factors of ecosystem service provision.

Another field is the acceptance by the potential applicants. Spatial and regional planning authorities, for example must comply with a large number of national and international regulations and laws. In Chap. 29, the potential as well as the limits of the ecosystem service assessment approach as a tool for balanced decision-making is analyzed.

The third field of the "proof of concept" in this chapter is dedicated to deal with the human dimension. Adding societal demands—economic as well as ethical ones—will shift weighting of individual ecosystem services, because ESS are still an anthropocentric approach. To do this weighting in a sensible way, requires application of the concept of sustainability. Chapter 30 is analyzing this field of problems.

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Mechanisms of Ecosystem Service Production: An Outcome of Ecosystem Functions and Ecological Integrity in Coastal Lagoons

28

Irmgard Blindow, Stefan Forster, Hendrik Schubert, Rhena Schumann, and Felix Müller

Abstract

Coastal lagoons provide important ecosystem services, but are simultaneously highly vulnerable. We aim at a better understanding of the mechanisms of ecosystem service production in these ecosystems. Three case studies, based on results obtained during the BACOSA and SECOS projects, identify the impact of the functional organism groups bioturbating zoobenthos, phytoplankton and macrophytes on coastal lagoons. These empirical results are merged with a theoretical framework on the relations between ecological conditions and ecosystem services, consisting of an integrative matrix projection. RESPON (relative ecosystem service potential) points are estimated for the three case studies. All functional groups have an overall positive effect on ecosystem services, and a very high impact on integrity parameters such as biodiversity, trophic efficiency and nutrient retention. The highest scores are obtained for macrophytes, while

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phytoplankton only has a slightly positive impact. For bioturbation, a major lack of knowledge was identified; bioturbating zoobenthos with high biodiversity is assumed to favour "seafloor integrity". Despite major difficulties such as lack of knowledge and highly different approaches, our analysis results in specific recommendations for management and future research. Management must consider the high connectivity of coastal lagoons with other ecosystems. Harsh impacts destroying benthic fauna communities have to be minimized. The promotion of submerged vegetation, which is an important provider of ecosystem services, must be implemented in the management of coastal lagoons.

28.1 Introduction

Since the growth phase of the ecosystem service (ES) concept has started by the end of the last Millennium, there has been the central question *"which are the mechanisms of ecosystem service production?"*, which honestly has not been answered satisfactorily till today for many ES and ecosystem types. One reason may be the enormous complexity, which surrounds the ecosystem service idea in human-environmental systems. Also the differences between the numerous single services do not support an easy comprehension. And additionally, the distinctions between scientific concepts in different disciplines may impede fast answers to this strongly interdisciplinary question. Although the problem of understanding the interactions between ecosystem structures, functions and services has been investigated from several aspects in the past (Barbier et al. 2011; Harrison et al. 2014; Liquete et al. 2016; Maes et al. 2016; Pascual et al. 2016; Erhard et al. 2017; Roche and Campagne 2017; Rodrigues et al. 2017; Grizzetti et al. 2019; Hammerschlag et al. 2019; Rullens et al. 2019; Teixeira et al. 2019), many questions are still unanswered.

Therefore, we try to illuminate some related aspects of this problem area for the investigated marine—coastal ecosystems: How can we connect ecosystem services and the empirical, ecosystem-based results achieved during the BACOSA project (see Chaps. 11, 12, 13, 18, this volume) to better understand the complex relations between ecological conditions and ecosystem services? This demand leads to more detailed questions, which are elaborated within this chapter:

(a) How can we better understand the production of ecosystem services on the base of intensive ecosystem research activities (case studies)?
(b) Which management- and research-related recommendations can be formulated based on these results?

In order to find answers to these questions, we have carried out three evaluative "thought experiments" based on the results of the BACOSA analyses of different coastal lagoons, thereby analyzing the impacts of three functional organism groups (bioturbating zoobenthos, submerged macrophytes and phytoplankton) on ES production.

These case studies are used to discuss the focal questions on the relations between ecosystem service production and dominating organisms of coastal lagoons. Consequently, this chapter was structured into a short description of the theoretical framework, some information on the methods and explanations of the three functional case studies. Subsequently, they are merged towards the focus of the ecosystem service approach, the outcome is discussed and some conclusions are drawn, thereby considering the impact of environmental conditions on species composition. The results of empirical ecological studies in coastal water bodies of the Southern Baltic Sea are combined with the outcomes of the ecosystem service studies. Thereby, highly quantitative results are linked with rather qualitative assessment strategies. This highlights a number of conceptual uncertainties, which are discussed, but still allows to draw major conclusions for both future research and management, which are presented at the end of the chapter.

28.2 The Theoretical Framework

The methodological starting points of this exercise were (i) the basic knowledge of ecological functions in coastal ecosystems, (ii) the results of six years research in the BACOSA and SECOS projects, (iii) long-term experience in regional analyses of the research area and (iv) recent theoretical ideas on the relations between ecosystem conditions and services (see e.g. Kandziora et al. 2013; Schneiders and Müller 2017). This last point can generally be described by the concept of the ecosystem service cascade (Haines-Young and Potschin 2010). Following the Chaps. 2 and 6 of this volume, ecosystem structures and processes are aggregated to the class of ecosystem functions, which have certain capacities to provide ecosystem services. The respective contributions to human welfare—the ecosystem services themselves—support benefits to humans and are therefore valued positively by the society. Consequently, the questions of this chapter are excerpts from a comprehensive network of interrelations in human-environmental systems; they focus on the biological and ecological relations that combine biotic and abiotic processors into functions, and value procedures of deriving services from these functions. Thereby, the role of biodiversity for the provision of ecosystem services is an important question.

Figure 28.1 highlights some of these aspects following Schneiders and Müller (2017): Self-organized ecosystem interactions form ecosystemic process bundles (e.g. carbon flows, nutrient flows), which link biotic active life-supporting processes to abiotic gradient structures. These processual components are integrated at the level of functions. The single features can be indicated by ecological integrity parameters, which include ecological process bundles, reflecting important carriers of ecosystem resilience and development (Müller et al. 2016). Furthermore, these processes are the basis for ecosystem service supply capacities and often summarized by the term ecosystem condition (Maes et al. 2016, 2018; Rendon

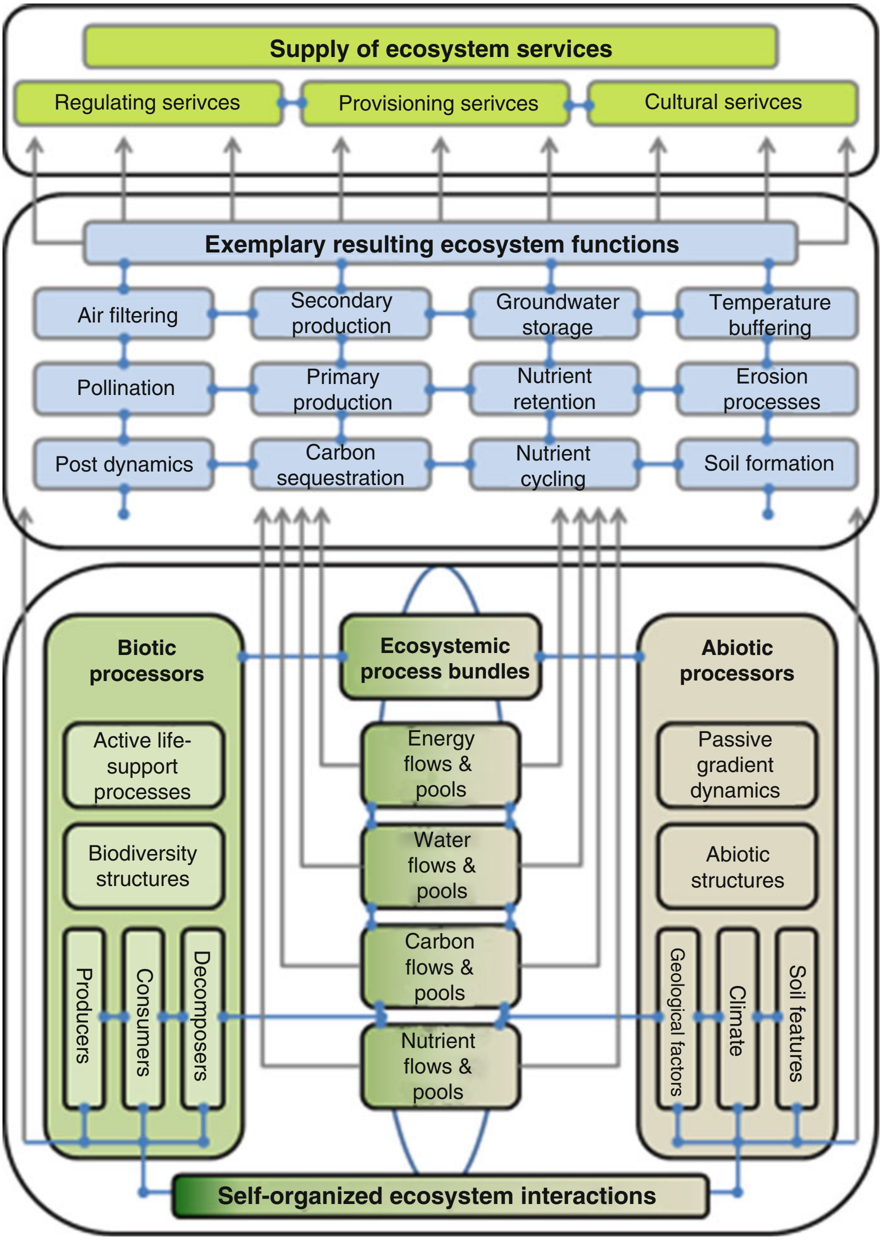

Fig. 28.1 Basic model of ecosystem service production referring to the cascade model of Haines-Young and Potschin (2010) after Kandziora et al. (2013) and Schneiders and Müller (2017)

et al. 2019). Figure 28.1 shows some of these components. On the third level of this figure, different functional components are combined with selected single processes to construct direct influence clusters towards human well-being. Thereby, each service is produced by distinct ecological components. Consequently, the ecological derivation of ecosystem service potentials turns out to be a very arbitrary, utility-focused selection mechanism. In contrast to the ecological integrity parameters, ES do not provide a holistic representation of the system, but are strongly concentrated on processes which support human welfare.

28.3 Methodological Starting Point

The following abstract assessments are based on the quantitative investigations of the projects SECOS and BACOSA as well as long-term investigations in the research area presented in this book (see Chap. 4).

Also the ecosystem service methodology was developed within the SECOS and BACOSA projects. As an outcome of multiple ecosystem service assessment approaches (see Chaps. 7 and 8 of this volume), an integrative ecosystem service matrix assessment was developed, which was applied to terrestrial, coastal and marine ecosystems (Schumacher et al. this volume; Müller et al. 2020; Bicking and Müller 2019; Burkhard et al. 2014). This matrix assesses the capacities of different ecosystem types to provide different ecosystem services. The resulting scoring system is based on an expert-guided relative assessment of ecosystem service potentials (RESPON) with basic values between 0 (no potential) and 100 (very high potential). The scores were derived from direct and indirect measurements (e.g. Kroll et al. 2012), expert assessments (e.g. Burkhard et al. 2009), regional statistics (e.g. Bicking et al. 2018), field tests (e.g. Stoll et al. 2015) and modelling results (e.g. Bicking et al. 2019), see also Chap. 24 and Müller et al. (2020).

In the following three case studies, we present the impact of three functional organism groups on the ecosystem service potentials of inner coastal ecosystems, based upon the data acquired from the investigation areas Darß-Zingst Bodden Chain (DZBC) and Vitter Bodden (VB) (see Chap. 4), long-term ecological data from the DZBC, and knowledge from a number of coastal lagoons.

28.4 The Case Studies

28.4.1 Case Study I: Bioturbation

The term "bioturbation" addresses "all transport processes carried out by animals that directly or indirectly affect sediment matrices" (Kristensen et al. 2012, p. 285), including transport of particles and solutes within the sediment and across the sediment–water interface (Fig. 28.2). This transport is mainly driven by benthic infauna activities, e.g. sorting of sediments for food particles, burrow construction

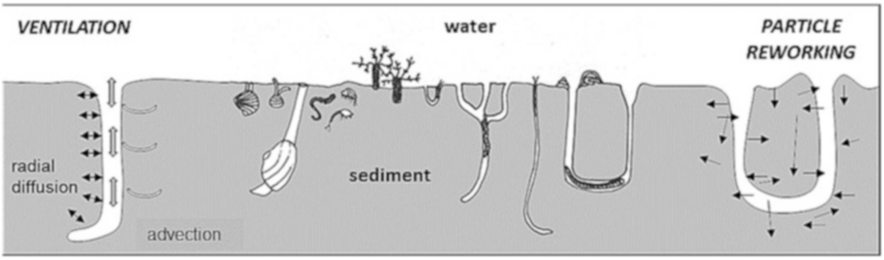

Fig. 28.2 Schematic illustration of bioturbating animals: major processes leading to fluid and particle transport in italics; arrows indicate the directions of biologically driven exchange processes. Processes illustrated: radial diffusion and pore water advection along burrows, random particle displacement during digging and maintenance of burrows. (based on an unpublished sketch by J. Renz)

and burrow ventilation. Transport by bioturbation frequently dominates over physical transport processes in marine environments shallower than 1000 m. Therefore, it is considered important for benthic–pelagic exchange and early diagenesis.

Marine scientists generally regard bioturbation as important for supporting sea-floor integrity (descriptor 6 in MSFD[1]) and the well-functioning of benthic ecosystems (Smith et al. 2016). Bioturbating organisms build up structures, affect the flow of matter and energy, and therefore shape "process bundles" (compare Fig. 28.1) integral to the way soft bottom aquatic ecosystems function and supply services. An illustrative way of looking at this effect is to ask what would be different if there was no bioturbating fauna. Today some anoxic deepwater areas in, e.g. the Black Sea and the Baltic Sea, display constrained but permanently anoxic sediments. Here bacterial life thrives, but no multicellular organisms, similar to times prior to the Cambrian Explosion some 500 million years ago. Under such circumstances, the material deposited on the sea floor forms undisturbed laminated sediments, where carbon preservation tends to be higher than in situations when oxygen is available and bioturbation occurs (Canfield 1994; Bockelmann et al. 2007).

Benthic animals function as "*ecosystem engineers*" that facilitate the occurrence of other species enhancing diversity (Jones et al. 1994). Structures like burrows and tubes created in the sediment are conduits of O_2 injected into largely anoxic sediments, thereby changing redox conditions. At the sediment-water interface mounds and tubes interact with water flow to exchange solutes and particles with the overlying water (Huettel et al. 1996). Surface structures also affect erosion and deposition at the sea floor (Friedrichs et al. 2009). Thus bioturbation enhances diagenetic processes and element cycling in most soft sediment ecosystems, diversifying nitrogen cycling (Aller and Aller 1998; Laverock et al. 2013), immobilizing phosphate, and affecting sulfur-, iron- and manganese cycling (van

[1] "Sea-floor integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected" according to https://ec.europa.eu/environment/marine/good-environmental-status/descriptor-6/index_en.htm

de Velde and Meysman 2016). In this context burial of many compounds decreases, but CO_2 liberation and transfer of energy into the food chain increase.

Bioturbation, however, is not one single or uniform process, which we could relate to ecosystem services in a simple or general manner. Brittle stars move sediment grains by pushing them laterally for short distances along the sediment surface, while the sand piper *Arenicola marina* moves grains more than 20 cm vertically within the sediment. Many organisms discriminate among particles according to their size or in search of food (Wheatcroft 1992; Graf 1992; Suchanek 1985; Gebhardt and Forster 2018). This makes particle transport selective, an aspect only marginally captured by current classifications of fauna into traits of bioturbation (François et al. 1997). Time scales associated with particle reworking and fluid pumping vary considerably with major consequences for associated meiofauna and bacteria at burrows, as redox conditions in the sediment fluctuate (Aller 1994; Forster 1996; Volkenborn et al. 2010). Differences in time scales and mechanisms of bioturbation affect reactions of sediment compounds in different ways and therefore may yield different effects. The link between bioturbation (Fig. 28.2) and any particular ES function beyond "integrity of the seafloor" is therefore not easily predicted. The causal link to a specific geochemical or biological effect may be understudied at present; in any case, these links are frequently context-dependent and non-linear in their relation.

An investigation with the flame retardant BDE-99 and cadmium (Hedman et al. 2008) demonstrated how physicochemical characteristics of the pollutant, burrowing depth and burrowing type as well as sedimentary organic matter interact to generate effects of burial versus mobilization. Bioturbation may trigger opposing effects, particularly when material reaches the so-called burial depth and is removed from ecological cycles for longer periods. Mixing exposes reactive fresh particle surfaces that support the adsorption of metals and organic pollutants, but is strongly dependent on the active biological species (Kristensen et al. 2011; Banta and Andersen 2003). As a result, some sediments may become sinks for pollutants. Conversely, with oxygen transport by fauna into the sediment leading to more mobile oxidized heavy metal compounds compared to sulfidic immobilization, sediments may become pollutant sources (Kersten 1988; Förstner and Salomons 1988; Hedman et al. 2008). Mixing of fresh and refractory carbon sources stimulates overall carbon degradation (Kristensen and Holmer 2001) and therefore CO_2—liberation from the sediments. Also the degradation of oil products is more efficient when infauna pump O_2-rich water into sediments (Christensen et al. 2002, Timmermann et al. 2002, 2003; Banta and Andersen 2003; Gilbert et al. 2001, 2003).

Generally, enhanced oxidation of sediments (by bioturbation) results in immobilization of soluble phosphate (PO_4^{3-}) through adsorption to particles (Forster and Bitschofsky 2015; Bitschofsky et al. 2015; Bonaglia et al. 2013; Thoms et al. 2018; Karlson et al. 2005), which counteracts a negative feedback between hypoxia in sediments and water column primary production (Conley et al. 2002). Some investigations show, however, that phosphate may be pumped from deeper anoxic sediments to the overlying water, if sufficiently deep-burrowing tube dwelling animals are abundant (Thoms et al. 2018; Renz and Forster 2014). Similarly, tube dwellers may stimulate denitrification to gaseous N_2, a process with remediation potential counteracting eutrophication. While this is evident from modelling and laboratory experiments of mostly single species (Pelegri and Blackburn 1996; Pelegri et al. 1994; Gilbert et al. 2003), it has been infrequently found in field studies, where species composition and abundance vary (Deutsch et al. 2010; Tuominen et al. 1998). The alternative bacterial pathway leading from nitrate to ammonia (DNRA), which is largely irrigated back to the overlying water, occurs in less oxidized sediments and retains N as ammonia in the system. In this case, there is only a small abatement effect on eutrophication (Karlson et al. 2005; Bonaglia et al. 2013).

Beyond the results from many specific bioturbation studies, our current knowledge suggests that mainly the degree to which animals increase the state of oxidation of a sediment (redox state) regulates net exchanges of N and P with the water column. Apart from burrow geometry, several regulating factors and their spatial and temporal dynamics are yet insufficiently understood. Interactions with bacterial performance, species- or trait-specific effects and the dependence of ES on density and composition of macrofauna communities determine the overall effects of bioturbation. Bioturbation is important in generating functions and ES. While there is a need for more research to understand how these services emerge under specific conditions, most ecosystem functions and ecosystem service production in this context are clearly related to the integrity of the benthic ecosystem. With respect to the demands of environmental practice, we utilized the recent cognition on bioturbation processes in order to assess their potential influences on the capacity of ecosystem service supply, realizing the multiple related causes of uncertainties.

28.4.2 Case Study II: Macrophytes

Submerged macrophytes have a “key function“in shallow aquatic ecosystems. Due to a number of feedback mechanisms, they increase water clarity, retain nutrients, thereby causing a reduction of phytoplankton densities, store carbon, and offer food, substrate and shelter for a number of organisms, including microalgae, zooplankton, macroinvertebrates, fish and waterfowl (Scheffer et al. 1993; Blindow et al. 2014; see Chap. 13).

In shallow aquatic ecosystems, submerged macrophytes therefore offer a number of support mechanisms for ecosystem service production. Enhanced water clarity and lower phytoplankton densities, including a reduction of toxic cyanobacteria blooms, improve the water quality and enhance the suitability of the ecosystem for touristic utilization, especially bathing. Both high availability of plant and macroinvertebrate food increase the ecosystem’s attractiveness for waterfowl (Milberg et al. 2002). Combined with enhanced water clarity, high densities of zooplankton and macroinvertebrates in areas with dense submerged vegetation improve predation efficiency and growth rates of fish (Persson and Crowder 1998; Hargeby et al. 2005). Additionally, submerged vegetation serves as reproduction habitat for fish. In the Greifswalder Bodden, the recruitment of herring has been assumed to have decreased due to the collapse of submerged vegetation (Kanstinger et al. 2016).

Our investigations in the intensively studied shallow lagoons VB and DZBC confirm this importance of submerged macrophytes (see Chap. 13). In spite of lower nutrient concentrations in the VB, total system net photosynthesis rates are far higher. Additionally, ecological transfer rates are far higher in this macrophyte-dominated system (Paar et al. 2021). Both higher net photosynthesis and higher trophic efficiency explain that ecosystem production is far higher in all trophic levels, including organisms that are of interest for human nutrition or recreation, such as fish and waterfowl (see Table 28.1), compared to the more nutrient-rich DZBC. Such a "paradox of enrichment" has for the first time been shown for shallow coastal ecosystems (Paar et al. 2021; see Chap. 13).

Transitions from a macrophyte-dominated to a phytoplankton-dominated state are thus crucial for ecosystem services. Unfortunately, such transitions are hard to predict due to a non-linear response of shallow aquatic ecosystems to external impacts such as changes in nutrient loading. Our investigations during the BACOSA project support the assumption that the shallow coastal lagoons of the Baltic Sea occur in two possible "alternative stable states", one of which characterized by clearwater and abundant submerged vegetation, the other characterized by phytoplankton dominance and turbid water (Meyer et al. 2019; Chap. 13). While the DZBC has been in a turbid state since a decrease of the submerged vegetation in the 1970s and 1980s (Walter 1973; Behrens 1982; Chap. 12), the VB is still dominated by dense submerged vegetation. A number of factors, however, indicate that this system is close to a so-called "tipping point", where small external disturbances may cause a "switch" and therefore, have a major impact on ecosystem conditions and services.

Ecosystems dominated by macrophytes have been shown to efficiently retain nutrients and store carbon. Coastal lagoons with a rich macrophyte vegetation therefore have an important function as filters between terrestrial (mainly anthropogenic) inputs and the open Baltic Sea (Asmala et al. 2019; Carstensen et al. 2020). In the investigated region, this function has been deteriorating substantially during the last decennia, due to a decrease of submerged vegetation caused by eutrophication. The DZBC and other estuarine lagoons have already lost their former rich macrophyte vegetation. Though these lagoons still retain a major part of the external nutrient input due to geomorphological and hydrographic conditions (Lampe et al. 2013), their filtering capacity is assumed to be far lower due to the short life span, high metabolism and elemental content of phytoplankton, which enhances turnover rates of carbon and nutrients (Villnäs et al. 2019). The outer marine lagoon VB is still in a macrophyte-dominated state. A change in species composition has occurred, however, from small, "bottom-dwelling" plants such as charophytes, to tall "canopy-formers" which retain nutrients less efficiently (Blindow et al. 2014, 2016). Together with different indications of high system variability and instability (see Chap. 13), this suggests that also the filtering capacity of this ecosystem already has decreased (Fig. 28.3).

The results obtained during the BACOSA project support earlier investigations which show that submerged macrophytes have a substantial positive impact on all

Table 28.1 Expert valuations (I) of the ecosystem service potentials (RESPON values) and spans of the expert assessments (II) of bioturbation, macrophytes and phytoplankton

		(I) Mean relative RESPON scores due to the impacts of (A), (B) and (C)			(II) Wingspan between minimum and maximum assessments of experts in (I)		
	Indicators	(A) Bioturbation	(B) Macrophytes	(C) Phytoplankton	(D) Bio-turbation	(E) Macro-phytes	(F) Phyto-plankton
Integrity attributes	Abiotic heterogeneity	24	25	3	10	10	20
	Biodiversity	25	29	2	5	5	25
	Ecosystem net primary production	-1	20	14	25	15	20
	Nutrient retention	13	18	9	20	20	40
	Trophic efficiency	17	22	4	20	15	40
	Storage capacity	8	18	6	40	20	10
Provisioning services	Crops (human nutrition)	3	3	0	10	10	10
	Biomass for energy	1	7	2	5	10	10
	Crops (fodder), aquaculture	3	4	0	5	10	0
	Livestock	4	3	2	5	10	10
	Fibres	1	2	0	5	20	0
	Wild food	11	8	7	20	25	15
	Fish and Seafood	12	16	6	25	30	10
	Flotsam	0	7	1	1	45	20
	Ornamentals	0	2	0	0	10	0
	Abiotic energy	0	0	0	0	0	0
	Minerals	0	0	0	0	0	0
Regulating services	Local climate regulation	0	2	0	0	10	0
	Global climate regulation	1	13	13	45	30	15

	Flood protection	-2	7	0	10	30	0
	Air quality regulation	0	0	0	0	0	0
	Erosion regulation, water	1	17	1	25	10	5
	Nutrient regulation	26	20	1	10	20	15
	Water purification	6	15	2	30	30	10
	Pest and disease control	5	7	−5	10	20	20
	Pollination	0	2	0	0	10	0
Cultural services	Recreation and tourism	7	0	−14	10	40	25
	Seascape aesthetics	7	11	−5	10	20	10
	Knowledge systems	7	6	5	10	20	10
	Cultural heritage	2	6	5	10	10	10
	Regional identity	2	6	0	10	20	0
	Natural heritage	14	14	2	10	20	10
	Sums:	197	308	60	386	535	360
	Average values:	6,2	9,6	1,9	13,1	16,7	11,8

The highest RESPON scores for every parameter are marked blue

Values indicate the size of deviation for the case studies A, B, C from the average values in the original assessment matrix (see Müller et al. 2020; Schuhmacher, this volume), where the overall potential has been characterized by scores between 0 and 100. A value of 20 means that the average value for coastal aquatic systems is enlarged by 20 points under the respective scenario conditions

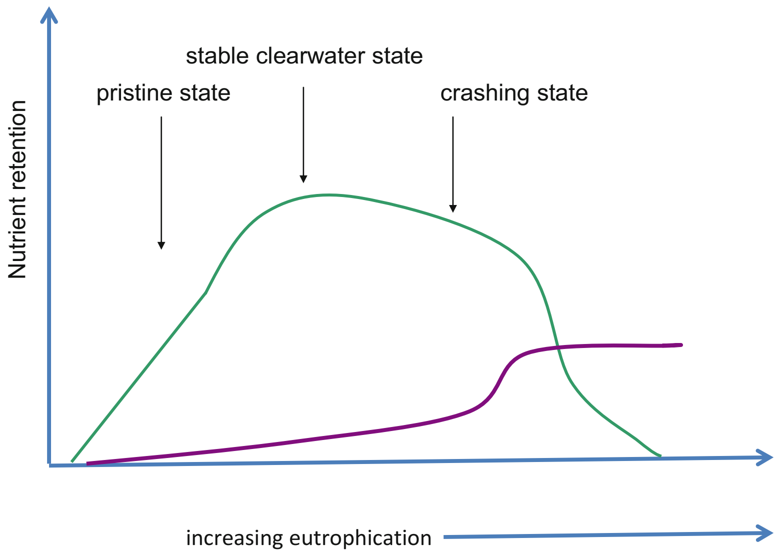

Fig. 28.3 Assumed nutrient retention by macrophytes (green line) and phytoplankton (violet line) in a eutrophication gradient. Note that in the crashing state, macrophyte biomasses are high, but the vegetation period is shortened causing a decrease in nutrient retention

integrity attributes. They provide a complex, three-dimensional structure with high biomass and abiotic heterogeneity, which stores substantial amounts of nutrients and carbon and forms the basis of a complex, species-rich food web with high trophic efficiency. We assume a positive impact on the provisioning ecosystem service “fish and seafood production”. Results obtained during the BACOSA project show higher trophic efficiency in all trophic levels in the macrophyte-dominated system, and higher growth rates of perch, a commercially important piscivorous fish. Among regulating services, strong impact is assumed on nutrient regulation and water purification. Contrasting impacts are assumed on the cultural service “recreation”: While submerged vegetation has a distinct positive effect on water quality, dense vegetation may impede activities such as boating, wind-surfing and bathing. A positive effect is expected on bird-watching.

28.4.3 Case Study III: Phytoplankton

Phytoplankton supports and generates many ecosystem services. As the main contributor to primary production in most aquatic ecosystems, phytoplankton produces oxygen and provides food for zooplankton and zoobenthos. Phytoplankton stores and retains nutrients, and increases energy transfer to higher trophic levels (Schubert 1984).

While these positive effects of phytoplankton are mainly observed/described at low or moderate nutrient conditions, eutrophication causes an increase in

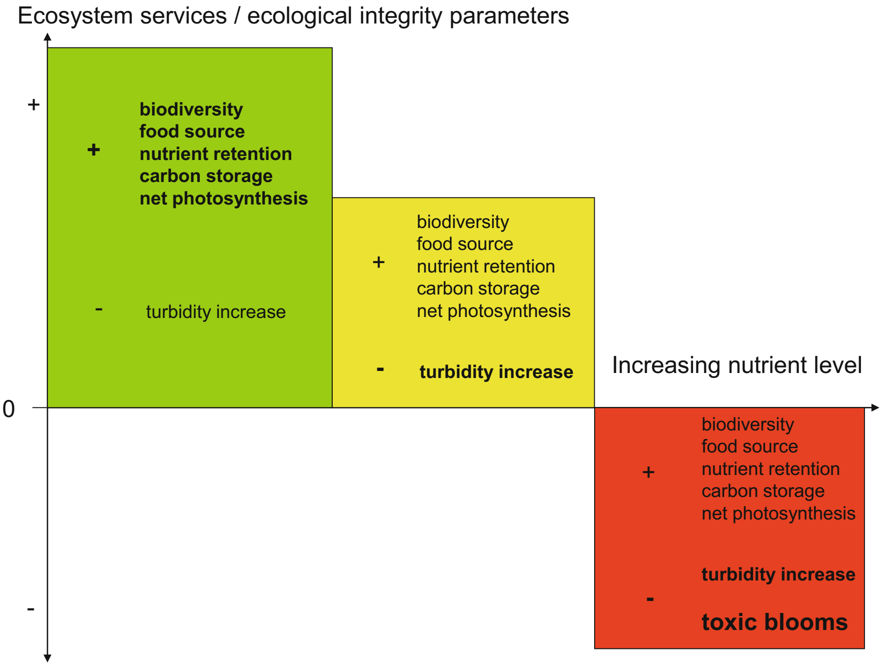

Fig. 28.4 Impact (relative scale) of phytoplankton on ecosystem services and integrity parameters along a eutrophication gradient. Strong impacts are indicated in bold. Note that two different phytoplankton conditions can be distinguished in highly eutrophicated ecosystems depending on absence/presence of toxic species

phytoplankton biomass, but often dominance of one or few taxa and thus a decrease of phytoplankton species richness (e.g. Bužančić et al. 2016). Eutrophication is also accompanied by an increase of negative effects from phytoplankton on ecosystem services and integrity parameters, which therefore can become negative in highly eutrophicated ecosystems (Fig. 28.4) Such "disservices" have also been described for other ecosystems (Dunn 2010; Schaubroeck 2017). Higher turbidity causes a decline of submerged vegetation (see Chap. 13; Fig. 28.3), which reduces the ecosystem services provided by this vegetation (see above). Additional negative effects of increasing phytoplankton biomass may be a reduction of food web structures and decrease of niches, overall lower species richness and a lower nutrient retention, coupled with increasing self-shading (Paar et al. 2021).

Upon strong eutrophication, phytoplankton blooms develop, which can be toxic at dominance of certain species of cyanobacteria and dinoflagellates. These blooms are harmful to humans, directly by poisoned food sources, and by indirect negative impacts (Karjalainen et al. 2008). Toxic blooms can occur in almost all aquatic ecosystems. Blooms also cause a self-limitation of the depth-integrated phytoplankton production. The negative impact of this situation on the ecosystem depends on the specific conditions, ratios between phytoplankton and macrophytes, and the respective food web structures (see Chap. 13).

Phytoplankton impact on ES is highly depending on its composition and density, which in turn is mainly influenced by eutrophication. Phytoplankton has positive impacts on all ecosystem integrity parameters, such as biodiversity, nutrient retention, trophic efficiency and carbon storage capacity, but especially on system net primary production (Paar et al. 2021; see Chap. 12). Due to this high primary productivity, phytoplankton also contributes to the provisioning service "fish and seafood production" and, due to carbon dioxide assimilation, to the regulating service "global climate regulation". At high densities and especially during blooms, however, phytoplankton has a negative impact on the cultural services "recreation and tourism" and "seascape aesthetics". Decreasing water transparency is a matter of concern especially among tourists, as the water appears as "dirty" with a low suitability for bathing. Toxic blooms, decrease of predatory fish and oxygen depletion have serious impacts on ES of shallow coastal waters. Especially cyanobacterial blooms gain high public attentions, as shown by newspaper reports in most summers.

28.5 Merging the Case Studies and the Theoretical Framework

The analysis above shows that all three functional organism groups imply important boundary conditions for the integrity of the related ecosystems (see Müller 2005; Müller and Burkhard 2010; Müller et al. 2010, 2020; Haase et al. 2018), as well as potentials to provide important ecosystem services. Following the rules of the ecosystem service matrix approach that have been described above (see Schuhmacher et al. this volume, Müller et al. 2020), the authors have searched for correction values, which should be connected to the basic matrix data, if one of the three functional groups is dominant. The maximal influence was defined by the value of 30 positive or negative RESPON points (relative ecosystem service potential, an overall span between 0 [no potential] and 100 [maximum potential]) characterizing the impact of the functional groups.

Table 28.1 shows these consequences for the three investigated case studies. Among single ecological integrity attributes, the factors heterogeneity, biodiversity and trophic efficiency receive a strong support by bioturbation and by macrophytes, while phytoplankton only has a moderate effect on these state values. Here, only the amount of energy taken up by the system (net primary production) is strongly increased. Compared to the impacts on ecological integrity, the influence on the ecosystem service classes seems to be rather low. Only wild food, fish and seafood are supported by bioturbation and macrophytes, in a smaller amount also by a phytoplankton. Among the class of regulating services, large effects have been assessed for phytoplankton and macrophytes on global climate regulation potentials due to high photosynthesis rates. Nutrient regulation is mainly affected by bioturbation and macrophytes. Finally, the cultural services are profiting from bioturbation

Table 28.2 Average RESPON scores of the expert assessments

	Average RESPON score	Average standard deviation	Average span of expert assessments	Average degree of expert uncertainty (0–3)	Average span of expert uncertainty (0–3)
Bioturbation	6,2	6,8	12,1	1,0	1,3
Macrophytes	9,7	5,3	17,0	1,2	1,9
Phytoplankton	1,9	4,9	11,3	1,0	1,2

Table 28.3 Average RESPON scores of the expert assessments related to ecosystem service and indicator classes

	Bioturbation		Macrophytes		Phytoplankton	
	AVG RESPON score	AVG Span	AVG RESPON score	AVG Span	AVG RESPON score	AVG Span
Ecological integrity	14,3	20,0	22,0	14,2	6,3	25,8
Provisioning services	3,2	6,9	4,7	15,5	1,6	6,8
Regulating services	4,1	14,4	9,2	17,8	1,4	7,2
Cultural services	6,5	10,0	7,2	21,7	−1,2	10,8

and macrophytes, while at high phytoplankton densities, the attraction for recreation and the seascape aesthetics is strongly reduced.

The right part of Table 28.1 adds the respective uncertainties by comparing the spans of the answers, which generally are high. In the bioturbation scenario, especially storage capacity, global climate regulation and water purification have high spans. They are mainly related to the question, whether the activity of the bioturbators increases the flows into the sediment, or whether releases from the sediment into the water body are dominating. This partly reflects the fact that the identity of benthic species in conjunction with the chemical matter in question may indeed generate substantially different and even opposing results. With respect to macrophytes, the uncertainties are highest on the relations with fish and seafood, floatsam, climate regulation, flood protection, water purification, and recreation. Finally, in the phytoplankton scenario, the spans are somewhat smaller, culminating in context with nutrient regulation and trophic efficiency. Overall, the largest problems for the evaluators appeared in context of crops (because in the Baltic environment the consumption of algae is a very small flow), floatsam (because the beachwrack can also be comprehended as a disservice), and knowledge systems (because one can learn from any constellation).

Tables 28.2 and 28.3 summarize these results: The highest effect on the overall ecosystem service potential can be ascribed to macrophytes, while bioturbation delivers a medium overall support. Phytoplankton gives smaller services, in a severely eutrophicated ecosystem even causing negative changes. The uncertainties of the assessing four scientists with expertise in empirical ecological investigations or ecosystem services, all familiar with the results obtained during the single case studies, are similar for the three functional groups (Table 28.2). Among the consequences for different ecosystem service types (Table 28.3), all functional groups have strongest, and the most direct influences on the ecological integrity attributes. The bioturbation scenario also supports cultural services, while the smallest influence relates to the provisions. Macrophytes have some effects on regulations. In phytoplankton, the cultural services receive negative average values, due to the severe impacts in the eutrophicated situation.

Figure 28.5 depicts the average assessments of the ecosystem service potentials (RESPON values) provided by the three functional groups in front of the respective span widths. Both values are highest in the macrophyte scenario, qualifying this functional group as the most valuable providers of ecosystem services. Bioturbation seems to support services in general on a medium level, while the phytoplankton scenario delivers the smallest service potentials.

The spider diagram of the bioturbation case study demonstrates that high potential values arise concerning ecosystem structures (e.g. heterogeneity, biodiversity), fish, and nutrient regulations. The experts' disagreements (spans) show summits referring to storage, global climate regulation and water purification. Here the outcome strongly depends on the system's situation, whether it releases sediment containments to the water body or whether it buries nutrients and carbon available in the water body into the sediment.

The overall effect of the ecosystem service potentials provided, as discussed before, becomes visible in Fig. 28.6: Here the basic values have been taken from the matrix values as described by Schuhmacher et al. (in this volume). We have chosen the evaluations for the ecosystem type "Lagoons & Estuaries (1130 & 1150), WFD type B1/B2: non-vegetated clay & mud" and have then applied our case study scenario results to this basic data set. In the Figures, the resulting data for a lagoon ecosystem are combined with the RESPON values for bioturbation, macrophyte dominance and phytoplankton dominance, respectively. For all cases studies, Fig. 28.6 shows the initial value from the Schumacher-matrix, the result obtained by combining these values and the deviation data from this study and the respective span as a measure of the inherent uncertainty of the analysis. We can find several similarities due to the original basic data, with peaks at the positions of fish and seafood provisions, global climate regulation, water purification and a relatively high valued block of cultural services. The highest values appear in the macrophyte scenario, and the lowest are again visible in the phytoplankton case study. Here the addition of the scenario conditions even reduces some of the individual service assessments.

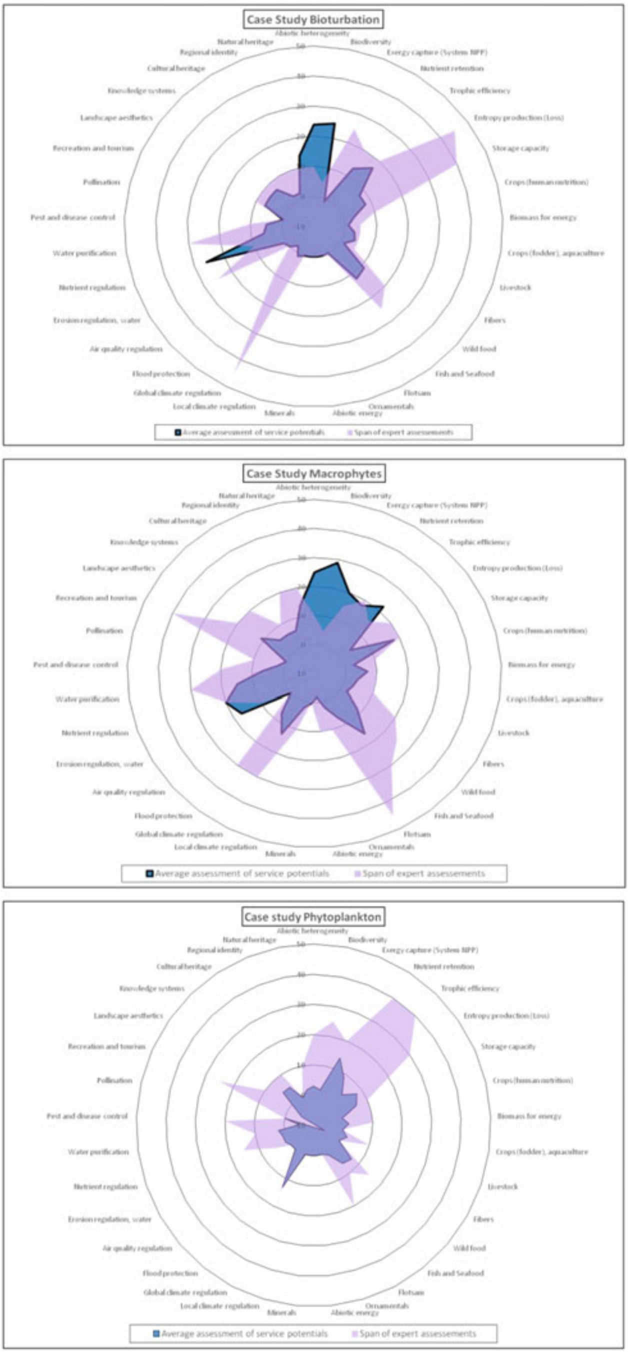

Fig. 28.5 Average relative ecosystem service potentials (RESPON values, blue) and span of expert assessments (pink) for the three case studies. The generalized influences were assessed within a data area from −30 (very strong reduction) to +30 (very strong increase)

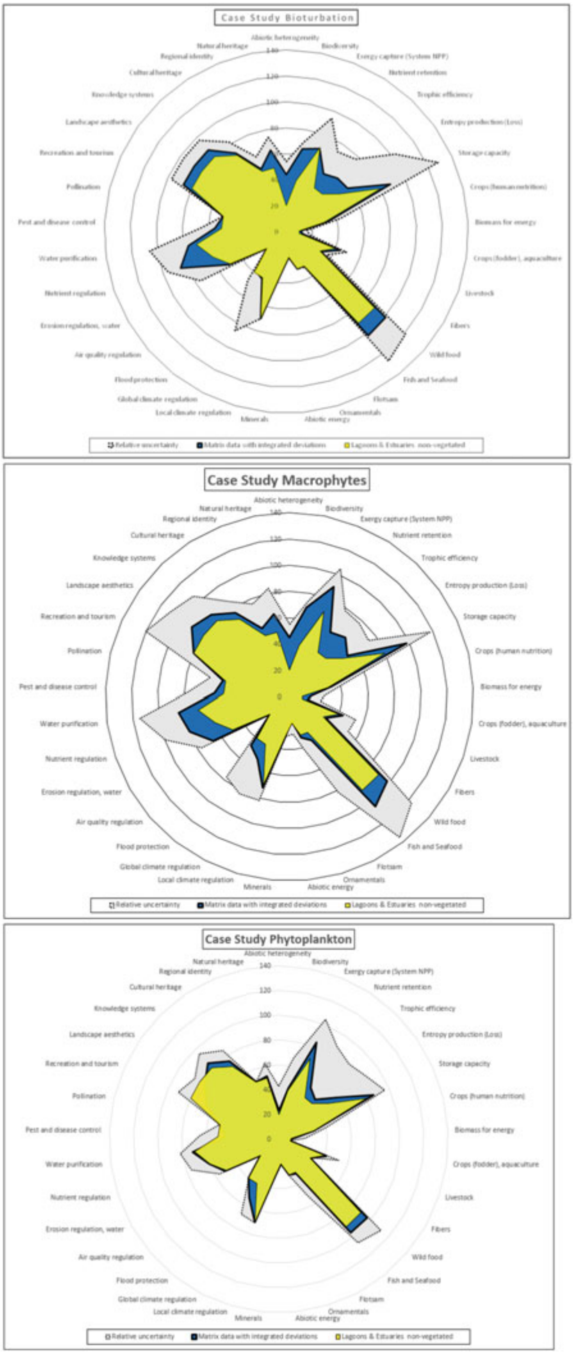

Fig. 28.6 Integrating the values for “Lagoons & Estuaries” from the matrix of Schumacher et al., (this volume, green) for non-vegetated lagoons and ecosystems with the average relative ecosystem service potentials (RESPON values, blue) and span of expert assessments (grey) for the three case studies

28.6 Discussion

28.6.1 Linking Empirical Ecologists to Ecosystem Service Specialists

In this chapter, we made an attempt to combine empirical ecosystem analysis with an expert-based ecosystem service assessment, in order to contribute to a joint understanding of ecosystem service production mechanisms. Three case studies were chosen, which had been analyzed in detail within the BACOSA and SECOS projects, as examples for intensively investigated, functionally important organism groups of coastal habitats. The necessary transformation from quantitative empirical analysis to more qualitative assessment procedures was accompanied with several productive outcomes, mainly for a better understanding of single services and for the recurring realization of complexity and locality—but also with methodological problems and interdisciplinary reservations. For the empiricist, the assessment techniques were filled up with unauthorized uncertainties on hardly walkable pathways, while the ecosystem service specialist side was wondering about the extraordinary demand for hesitations and discussions based on trifles and details. So we experienced a typical dispute between different degrees of reductionism and holism, fortunately ending in constructive emergent properties.

28.6.2 Linking Ecological Investigations to Ecosystem Service Production

All three case studies describe the intricate interactions between the environment and a "functional organism group" (bioturbating zoobenthos, submerged macrophytes and phytoplankton). As already shown in numerous investigations, all three functional groups affect the whole ecosystem to such an extent that ecosystem structure and functioning differ considerably depending on the abundance of this functional group. Such organism groups therefore have been called "key organisms" in ecological investigations (Goggina et al. 2017). Here, we present for the first time quantitative estimates of the major impact of such "key organisms" on the ES potentials in shallow coastal lagoons, supported by the high RESPON scores achieved in all three case studies.

Our investigations within the BACOSA project further show that there is not "the" coastal lagoon, but that single lagoons differ considerably in food web structure and functioning. Apart from "key organism" dominance patterns, hydrological characteristics and anthropogenic impact, especially eutrophication is responsible for these differences (see Chap. 4). Consequently, ES potentials differ considerably among single coastal lagoons.

Ecological interactions are intricate. Abiotic and biotic components are interlinked in a web-like pattern of mutual interrelationships. Anthropogenic impact has a major influence on dominance patterns of organism groups, including "key organisms", while "key organisms" are able to modify their abiotic "frame" conditions, often to a substantial extent. Further, a direct transfer from ecological characteristics to ES is not possible. Finally, ecosystems including their "key organisms" are not only affected by anthropogenic impact, but react on and modify this impact, which has to be considered in management measures.

Thus, differences in methodologies and "languages" used by empirical ecologists and ecosystem service specialists were not the only challenge we had to face during this joint analysis—already the subject per se was all but trivial. In spite of all difficulties, we can draw some stimulating points for subsequent discussions, and finally, give some recommendations for management and future investigations of coastal lagoons.

Although data scarcity is a focal and recurrent starting point for scientific grousing and moaning, we have to realize that we are arguing from a rather luxury position. Concerning applied assessment, our case studies are good examples for interpolations within the matrix approach. The existing matrices (e.g. Burkhard et al. 2014; Müller et al. 2020; Schuhmacher et al. this volume) can depict probabilities for service supplies for a restricted number of ecosystems only. With the expert-based interpretation of the situations in the DZBC and the VB, it has become necessary to define further ecosystem types and to use variations of their structural features. In spite of several doubts, we could show that such an interpolation can be done, thereby increasing the applicability of the matrix approach extremely. This result is also valid with respect to functional units: On the one hand, we have learnt to distinguish the outcomes of bioturbation; on the other hand, the sequences and processes of eutrophication were applied to demonstrate the consequences of functional ecosystem shifts. Based on such experience, also scenarios are applicable.

28.6.3 The Role of Biodiversity

Generally, "biotic processors" (see Fig. 28.1) strongly influence the outcome of functional interaction of ecosystem processes. Specifically, we could not only assess the important roles for ES generation of the "key organisms" investigated, but also show that these roles vary depending on the taxonomic composition and species richness of these functional traits. Thus, not only the density, but also the taxonomic composition of phytoplankton decides upon the delivery of services versus disservices (see Fig. 28.4). Bioturbating organisms can both reduce and enhance carbon sequestration and nutrient storage depending on their taxonomic composition. Different life forms of macrophytes dominate at different trophic states, including lower stability and a weakening of the ES provided by this community at higher nutrient concentrations (see Chap. 13). As a general observation, functional groups with high biodiversity have been shown to provide higher ecosystem stability (Naeem and Wright 2003; Cardinale et al. 2006). Biodiversity is playing a prominent role in ecosystem functioning and consequently, in the production of ecosystem services. Biodiversity was therefore regarded as an important ecosystem service, with some direct impact on cultural services (ecotourism, bird-watching etc.). Biodiversity further has a considerable, but indirect importance for human welfare, via its high impact on integrity indicators.

28.6.4 The Role of Dynamic Changes

As illustrated in detail in Chap. 13, ecosystems do not react linearly on external (including anthropogenic) influences. Dominating functional groups, and especially "key organisms", are able to counteract and "buffer" such impact by a number of feedback mechanisms. Along a gradual change of external conditions such as eutrophication, the ecosystem therefore first seems "unaffected" until a certain threshold is passed and it switches across its "tipping point" (see Chap. 13). Close to the "tipping point, small external disturbances can cause a major, often unexpected change in both abiotic parameters and food web composition, and a similarly substantial change in ES supply.

28.6.5 The Role of Distinct Viewpoints

ES were developed as a tool for comprehensive valuation of all aspects of human-ecosystem relations. Not only different groups of stakeholders, however, disagree in their assessment of specific services, but also experts with a comparable background differed largely in their assessment even of services directly linked to their field of expertise. For some ecology integrity components such as system net primary production or ES such as biomass for energy, quantifiable estimates can be given, but especially cultural services are notoriously difficult to quantify, though empirical values contribute to the outcome also of these ES. To get a balanced valuation, a thorough discussion of the individual aspects of each service is necessary, as experienced in this study. A specific example illustrating the disagreement among experts is flotsam, a natural component of beach ecosystems and generated mainly by macrophytes. Flotsam is often seen as a nuisance by tourists and consequently, recreation resort managers, but may be a valuable resource, an important element for coastal protection by providing nutrients for dune colonization, and enhances biodiversity by providing habitat heterogeneity.

28.6.6 The Role of Scales

A specific characteristic of the investigated coastal ecosystems is their high connectivity. Water exchange rates among the lagoons and from terrestrial ecosystems to coastal lagoons and finally, to the open Baltic Sea are high, resulting in an export of services and disservices to adjacent regions. High nutrient retention in a coastal lagoon may prevent the adjacent open Baltic Sea from eutrophication, but may be accompanied with high phytoplankton densities and therefore reduced water transparency and cultural disservices, such as the lagoon's unsuitability for bathing during an algal bloom. A rewetted coastal peatland may have improved properties, like more diverse flora and fauna, carbon storage and flood protection, but releases phosphorus for decades (Zak and Gelbrecht 2007), which is a great disservice for adjacent coastal lagoons. Unfortunately, ES assessments of rewetted peatlands focus exclusively on functions within these ecosystems (Zerbe et al. 2013). Results from the DZBC illustrate the value and necessity of data with high spatial and temporal resolution. While narrow zones within the reedbelts can release high amounts of phosphorus (Karstens et al. 2015), any phosphorus release from the sediments in the major part of the lagoon could not be shown except for high, but short and rare release events during oxygen drops under ice cover (see Chap. 12). Recent observations that anoxic conditions within the sediment locally "reach out" into the water column (Karstens et al. 2015; Bochert pers. com., Schumann unpublished) raise the question whether the lagoon system in future will retain or release nutrients. The importance of temporal scales is described in detail in Chap. 12, which illustrates short-term and long-term changes of different parameters in the Darß-Zingst Bodden chain. Like other coastal lagoons, this ecosystem is exposed to irregular, short, but drastic water level changes and exchange with the open Baltic Sea. Chapter 12 shows that extreme events such as oxygen drops or drastic water exchange rates influence a number of parameters, e.g. nutrient concentrations and transparency and finally, ecosystem services for a long time period, and illustrates the importance of high-resoluted data sets to identify and quantify the impact of such events. The question of positive or more negative effects of eutrophication or some components of the lagoon, like phytoplankton, depends also on the point of view on system borders.

Ecosystem disservices provided by the highly loaded Bodden systems thus implies a high advantage for the conditions in the open Baltic Sea. Besides these scale distinctions, we also should consider that each ES-producing process is operating on an individual spatio-temporal scale with individual developments and reaction characteristics.

28.6.7 The Role of Uncertainties

There are many causes for uncertainties in such ES assessments. Their formation and their consequences have been discussed in several papers (e.g. Hou et al. 2013; Campagne et al. 2017, 2020; Campagne and Roche 2018) and some methodological consequences have been drawn (see Chap. 24 and Müller et al. 2020). Some of these uncertainties are based on facts beyond our knowledge of structure and functioning of ecosystems, and the interactions and controls generating ecosystem services. Four scientists with different specific expertise estimated/valued changes in ES in response of bioturbation, dominance of microalgae or macrophytes for each of these three scenarios. Their professional backgrounds clearly entail deviating judgments and uncertainty in judging scenarios other than one's own field of expertise. Given that scientists would probably prefer to judge based on facts we consider the uncertainties shown in Figs. 28.5 and 28.6 quite acceptable. Moreover we do not anticipate that these uncertainties will be easily/much reduced once more data are measured in ecosystem studies.

28.6.8 Connecting Ecosystem Services and Empirical, Ecosystem-Based Results

The focal question (see introduction) was related to the connection of ecosystem services and the empirical, ecosystem-based results achieved during the BACOSA project. We have chosen the expert-based matrix approach, and as a result the authors can state that it has been suitable to better understand the complex relations between ecosystem services and ecosystem conditions. Within this experiment, we have not focussed on one or two ecosystem services with good quantitative knowledge bases, but chosen a holistic approach with comprehensive ES bundles. Thereby, we had to accept a substantial gradient between empirical, rather detailed knowledge and systems-based uncertainties; looking at the whole system was in this case, however, more significant than quantifying further details. Applying this starting point, we have to state that a direct transfer of ecological data to integrity attributes (which ARE ecological variables) is possible, but very often data are lacking and mechanisms (impacts of key organisms) still rather unknown. Due to lacking data, it is not possible to give any reliable empirical numbers for nutrient retention or carbon storage of the coastal lagoons studied. Naturally, the derived service valuations cannot be more exact than the sketchy basic data.

In our analysis, we were also able to transfer empirical data into information on ecosystem services. Thereby, the integrity attributes served as "intermediate variables", as a direct transfer rarely was possible. Several provisioning ecosystem services, such as seafood, aquaculture production and bioenergy can be derived from empirical data such as production of different organism groups. Regulating ES can be derived and indicated basing upon the integrity indicators or ecological modelling results. Quantitative estimations are, however, encumbered with high uncertainties because of a substantial lack of empirical data. We could further estimate the impact of the "key organisms" on cultural ecosystem services, but only provide qualitative values. Summarizing, there are several methodological and strategic problems—although we have been working with an extraordinary data situation—far from the "normal" conditions of environmental decision making. Nevertheless, the answers to the initial queries of this chapter are optimistic:

(a) How can we understand the creation of ecosystem services better?
The case studies illustrate the intricate interrelationships between functional organism groups and ecosystems. In order to deduce at least (complete) ES bundles, not only individual processes should be studied. Different experimental approaches (e.g. Artificial Neural Networks, Bayesian ANOVA) may further help to analyze specific ecosystem functions. In spite of the upcoming difficulties, complex analyses are necessary. One example for such an approach is the application of various methods such as stable isotope analyses and ENA modelling to understand the functioning of the complex food webs of the DZBC and VB within the BACOSA project (see Chap. 13). Here, the expert knowledge achieved in this analysis has been used to transfer this ecological analysis into an identification and assessment of ES.

(b) Which management and research-related recommendations can be formulated based on these results?
ES provide a brilliant tool to illustrate what different aspects ecosystems have for the human society and to demonstrate the effects that human impacts may have on the provision of ES. This can be employed to prepare decisions about scenarios of ecosystem use, which balance the different stakeholders opinions. The demand for such applications of ES in management is steadily growing. Therefore, we need more information and valuation about more ecosystem types, stronger distinguished services, more experts who help to improve respective matrices, more case studies and real-life-applications, more elaborated tools. Approaches like the one described here have to be further developed. Due to the difficulties to transfer ecological characteristics to ES (see above), integrity indicators are a useful tool and indicator of ecosystem and stability. These indicators should therefore be in the focus of management recommendations, opening the door for an increased implementation of the ES approach.

Our case studies result in specific recommendations for management and future research. These recommendations also have to consider the high connectivity of coastal lagoons with other lagoons, but also terrestrial ecosystems and the open Baltic Sea. This connectivity is expressed in not only in high water exchange rates, and transport of nutrients and other abiotic matters across ecosystem boarders, but also in the migration of different organisms among the single habitats/ecosystems. Just one example is the fundamental function of coastal lagoons for recruitment of herring, one of the most important commercial fish of the Baltic Sea (Kanstinger et al. 2016). Polte et al. (2021) demonstrated that the timing of annual spawning periods has a major impact on the recruitment success of herring (*Clupea harengus*) in the western Baltic Sea. They assumed that the synergistic effect of climate change and eutrophication causes a severe pressure on fish early life stages. Our comparison between DZBC and VB shows that zooplankton densities are lower in the lagoon without macrophytes, especially in spring, and thus confirm that eutrophication of shallow lagoons might have a negative impact on fish recruitment. This emphasizes the need for cross-ecosystem management strategies (Eriksson et al. 2011; Reusch et al. 2018). If we look at our three case studies, we can summarize the following:

Bioturbation: Investigations indicate opposing impacts of bioturbation on fundamental ES such as nutrient retention and carbon storage, but many cause-effect relations are not clearly identified yet. Due to the "umbrella character" (Kristensen et al. 2012) of the term bioturbation we have difficulties inferring quantitative relations of some important processes, such as denitrification, to measured bioturbation intensity. Therefore, it is not possible to give management recommendations in these cases. It is not perceivable how technically a bioturbation community that provides positive services could be designed/created. Neither may it be desirable. Instead, a major need for future research can be identified. There should be more focus on cause-effect-relations, interactions and feedback mechanisms. There seems to be an agreement, however, that "seafloor integrity" (also an umbrella term) should be taken care of in order to allow a stable community of bioturbating zoobenthos with high biodiversity. For management this implies that harsh impacts destroying benthic fauna communities have to be minimized.

Submerged vegetation has since long been known to stabilize the clearwater state and to provide a number of ecosystem services in freshwater lakes. Lake restoration therefore aims at promoting this vegetation. The most important management tools applied are reduction of external nutrient supply, reduction of the internal nutrient pool, biomanipulation, and implementations of "wave-breakers", either as artificial structures or plantations of suitable submerged macrophytes. (Hilt et al. 2006, 2018).

For coastal lagoons, we show for the first time that submerged vegetation not only has a major impact on the whole food web, but also is an important provider of ecosystem services. We conclude that the promotion of this vegetation has to be implemented also in the management of these ecosystems. Increased nutrient loading has been identified as the reason for the disappearance of submerged vegetation in some coastal lagoons, while other lagoons still have dense submerged vegetation (see Chap. 4). A substantial reduction of these loads has been achieved since the 1990s, but has not yet caused any major re-colonization of the submerged vegetation in the DZBC and Greifswald Bodden (Munkes 2005; Paar et al. 2021). Further reductions of the nutrient loads increase the probability for a re-colonization, but the necessary amounts of reduction can not be quantified, as we do not know how close these ecosystems are to the "tipping point" (see Chap. 13). In contrast to freshwater ecosystems, only few investigations have identified and quantified the feedback mechanisms for either of the two states, and further studies are badly needed. Biomanipulation seems not to be a promising tool because of the large size of the coastal lagoons, and their openness and the high migration rates of fish along the Baltic Sea coast (Eklöf et al. 2012). Implementation of wave-breakers may be a suitable tool to increase the chances for macrophyte recovery in isolated parts of estuarine lagoons. Any plantations of submerged macrophytes, preferrably in sheltered bays and/or in sheltered enclosures, is promising only if the conditions are good enough to allow positive growth rates for these plants (Bakker et al. 2013). Because of the poor light availability, a successful colonization can be expected only in shallow (marginal) regions of the estuarine lagoons. An expansion to deeper water, which is necessary for the establishment of a clearwater state, is dependent on a longer period of favourable weather and hydrology conditions.

The VB, still in a favourable macrophyte-dominated state, may be close to the "tipping point" (see Chap. 13). Any negative impact that may cause a switch to the turbid state should be avoided, as a return to macrophyte dominance would then need major and cost-consuming actions.Most important is avoidance of any further nutrient increase. Digging and construction activities may cause increased turbidity and should be limited. Piscivorous fish, which has be shown to have a substantional impact on filamentous algae in coastal brackish lagoons (Donadi et al. 2017), is increasingly exploitated by commercial and especially, recreational fishery, and pike (*Esox lucius*) shows first signs of recruitment overfishing in our investigation area (van Gemert et al. 2021). A limitation of recreational fishery is therefore recommended.

In contrast to freshwater ecosystems, alternative state patterns in coastal lagoons are poorly investigated (see Chap. 13). Further studies are badly needed to predict regime shifts and to develop successful management strategies. Because of the non-linear response of shallow aquatic ecosystems, this is a challenging task, as large increases in the indicators only “occur once a regime shift already is initiated, often too late for management to avert it” (Biggs et al. 2009). Intensive research has recently aimed at detecting “early warning signals” of regime shifts, and identified increases in variance, increased system skewness and slow recovery after disturbances as possible indicators (van de Leemput et al. 2018).

Especially under low and moderate nutrient conditions, phytoplankton provides positive ecosystem services. In highly eutrophicated ecosystems, however, both integrity parameters and ecosystem service values decrease due to a shift in species composition to few, highly grazing-resistant taxa dominated by cyanobacteria. This causes lower trophic transfer efficiency and ultimately, an ecosystem with low production also of higher trophic levels including species used as seafood (see Chaps. 12 and 13). Apart from grazing resistance, a high nutrient efficiency causes a high stability of such cyanobacteria communities (see Chap. 12). In such ecosystems, also toxic algal blooms may develop, which can give rise even to negative ES values (see Fig. 28.4).

Management therefore should aim at controlling and reducing both internal and external nutrient loadings. Further, there is a high need for research on factors stabilizing and de-stabilizing grazing-resistent cyanobacteria communities.

28.7 Conclusions

In this chapter, we have tried to illuminate the coupling of an expert-based ecosystem service assessment and empirical, ecosystem-based knowledge to better understand the complex relations between ecosystem services and ecosystem conditions. This connection between deep processual knowledge on ecosystem structures and processes and usable, modern recipes for practical environmental management is a long bridge, whereby the connected islands can be rather distant from each other and the lanes may be quite instable, in some cases being pathways only. Nevertheless, we have to use this link in order to find long-term sustainable solutions on a suitable scientific basis. Therefore, the elaboration of applied concepts for ecosystem service management based upon ecosystem analysis will remain a very important task in future.

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Ecosystem Service Assessment in European Coastal and Marine Policies

29

Gerald Schernewski and Esther Robbe

Abstract

We provide an overview about the present state of the art and discuss the possibilities and limits of ecosystem service assessments in the implementation of various European coastal and marine policies, such as Marine Strategy Framework Directive, Water Framework Directive, Maritime Spatial Planning Directive, Habitats-Directive (Natura 2000), and Integrated Coastal Zone Management. Ecosystem Service approaches have many strengths, but usually also suffer from weaknesses, for example, a limited reliability, oversimplification, a heterogeneous approach, a weak scientific basis, a merely anthropocentric view on nature, and a very strong scale dependence. Despite these limitations, the ecosystem service concept can play an important role in policy implementation: as supporting approach to establish a link between humans and nature and introduce a socio-ecological-economic view on nature protection, preservation, and sustainable use; to support communication with and information to the public; to offer new possibilities for a mobilization and a guided, active involvement of stakeholders; to allow the comparison of different ecosystem states in the past, presence and future and across regions; and to enable a more comprehensive definition of targets and policy objectives. Further, it enables the comparison and prioritization of different environmental measures, conservation and restoration approaches or management concepts.

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29.1 Introduction

The marine and coastal environment in Europe is under severe pressure. Intensified human uses, climate change as well as ongoing land-based and sea-based pollution are threats for these ecosystems and valuable habitats. As consequence, the protection and sustainable development of coasts and marine waters became a focus of the European Union's environmental policy.

Important elements are the Marine Strategy Framework Directive (MSFD), or Marine Directive, the Water Framework Directive (WFD), the Maritime Spatial Planning (MSP) Directive, the Habitats-Directive (Natura 2000), the Invasive Alien Species (IAS) regulation, the Recommendations on Integrated Coastal Zone Management (ICZM) and the Biodiversity strategies 2020 and 2030.

The ecosystem services (ES) concept was not explicitly considered in EU environmental policy before 2008. The Marine Directive became the first EU policy containing the ES concept (Bouwma et al. 2018). Today, most of the coastal and marine EU policies address the ES concept and assessments directly or indirectly in supporting documents guiding the implementation process (Bouwma et al. 2018).

Usually it is expected that ES assessments provide a comprehensive understanding of structures and (inter-)dependencies between humans and the environment and that they support the required "Ecosystem Approach to Management" (Seppelt et al. 2012; Baker et al. 2013). However, ideas and recommendations about the concrete aims of an ES assessment are either lacking or remain vague, e.g., where and when to carry it out in the policy implementation process as well as how to use the results. ES assessments are based on different methods and differ in reliability and robustness (Seppelt et al. 2012). Especially the monetary valuation of ES remains a challenge (Mehvar et al. 2018). Further, Liquete et al. (2013a) point out the need of an improved ES classification that meets the specific requirements of marine and coastal systems and policies.

Our objectives are to provide an overview about the present state of the art and discuss the possibilities and limits of ES assessments in the implementation of various European coastal and marine policies. Our analysis focusses on examples from southern Baltic coastal systems, especially taking into account structures and functions of the inner coastal water body ecosystems.

29.2 Biodiversity Strategy and Habitats Directive

Adopted in 1992, the legally binding "Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora" (Habitats Directive) aims to promote the maintenance of biodiversity, taking account of economic, social, cultural, and regional requirements. Together with the Birds Directive, it forms a major element of Europe's nature conservation policy and establishes the EU-wide Natura 2000 ecological network of protected areas, safeguarded against potentially damaging developments.[1] Additionally, the Habitats Directive became a core element in implementing the EU Biodiversity Strategy.

In 2011, the EU adopted its non-binding Biodiversity Strategy setting out 6 targets and 20 actions to stop the loss of biodiversity and ES in the EU by 2020.[2] The targets are (1) to protect species and habitats (implemented by the Birds and Habitats Directives); (2) maintain and restore ecosystems; (3) achieve more sustainable agriculture and forestry; (4) make fishing more sustainable and seas healthier; (5) combat invasive alien species (supported by EU Regulation 1143/2014 on Invasive Alien Species); (6) help stop the loss of global biodiversity.

Target 2 of the Strategy, aiming to maintain and enhance ecosystems and their services, includes several actions. Some incorporate instruments for integrating ecosystems and their services into decision-making. In Action 5, the Member States are requested to map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by 2020. Action 6 aims to restore degraded ecosystems and their services in the EU and Action 7 aims to ensure no net loss of biodiversity and ES.

Most studies specifically focusing on the ES concept within the Biodiversity Strategy and its implementation arose from the initiative and working group on Mapping and Assessment of Ecosystems and their Services (MAES). Maes et al. (2012) started to map ES (here water purification) for policy support and decision-making on the EU level. In 2013, the European Commission published a discussion paper specifically on an analytical framework for ecosystem assessments under Action 5 of the EU Biodiversity Strategy to 2020 (Maes et al. 2013). As MAES is one of the keystone actions of the EU Biodiversity Strategy to 2020 (Maes et al. 2015), they developed an analytical framework that supports the implementation of the EU Biodiversity Strategy to 2020 by a consistent set of indicators (Maes et al. 2016). Influenced by the MAES approach, the Common International Classification of Ecosystem Services (CICES) was developed (Haines-Young and Potschin 2013, 2018), which is the commonly used typology in recent European literature.

Studies consider the ES concept as a must within the Biodiversity strategy, as its strength lies in the need to consider the entire interlinked socio-economic-ecological system (Burghilă et al. 2016). Blasi et al. (2017) assume that ES assessment results may help decision-makers by providing a timely, accurate, and relevant contribution to knowledge of biodiversity. Further, many concrete examples exist how ES assessments can support the practical management of habitats, for example, reed management (Karstens et al. 2019).

Target 5 includes Action 16, aiming to identify invasive alien species (IAS), to control or eradicate priority species, and to manage pathways to prevent new invasive species from disrupting European biodiversity. To provide a legal framework to fight IAS the EU Regulation 1143/2014 on invasive alien species entered

[1] http://ec.europa.eu/environment/nature/legislation/habitatsdirective/index_en.htm

[2] http://ec.europa.eu/environment/nature/biodiversity/strategy/index_en.htm

into force in 2015. The IAS regulation was the first that explicitly promoted a definition of ES and integrated the ES concept within implementation on local level (Bouwma et al. 2018). Katsanevakis et al. (2014) analyze the positive and negative impacts of invasive alien marine species on ES and biodiversity, showing that the positive impacts are probably underestimated due to a perception bias.

Our current knowledge on both negative and positive impacts of invasive alien species on ecosystems and biodiversity is based on mostly qualitative data. Thus, environmental management decisions are often difficult and controversial due to the complexity of species interactions (Katsanevakis et al. 2014). ES assessments can support the quantification and mapping of the impacts of invasions building a better understanding of socio-ecological system interactions and thus, can assist managers and policy makers in their decisions on prevention or mitigation actions to be taken.

The mid-term review of the EU biodiversity strategy[3] concludes that progress with respect to maintain and restore ecosystems and their services (Target 2) is visible. However, it takes place at insufficient rates and efforts to combat invasive alien species (Target 5) are being implemented. Despite well-established conceptual ES frameworks and many case studies, shortcomings exist. The EU report on mapping and assessment of ecosystems and their services[4] provides an overview, however, the required full mapping and assessment of the state and economic value of ecosystems and their services in the entire EU territory is still underway. Comprehensive, applicable ES assessment concepts across the land-sea interface are still lacking, but are required to make different systems comparable and to enable a general application of ES concepts in Natura 2000 management.

Monetary ES assessments, as required in the BDS, are hampered by many methodological problems and limitations and hardly provide reliable information for decision-making. Additionally, Kuhn et al. (2021) identify existing gaps in ecosystem services research with focus on the Baltic Sea region.

29.3 European Water Framework Directive

The "Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for the Community action in the field of water policy," in short, the EU Water Framework Directive (WFD), was adopted in 2000. This framework legislation was meant to overcome fragmentation of water policy in the European Union. It had the aims to: expand water protection to surface waters and groundwater; achieve a good status for all waters until a defined deadline; carry out water management on river basins level; combine emission limit values and quality

[3] https://ec.europa.eu/environment/nature/biodiversity/comm2006/pdf/mid_term_review_summary.pdf

[4] https://biodiversity.europa.eu/ecosystems

standards; include cost and cost-effectiveness aspects; involve citizens and stakeholders and streamline legislation.[5]

Overall objectives are the protection of the aquatic ecology, the specific protection of unique and valuable habitats, the protection of drinking water resources, as well as the protection of bathing water. These objectives are all integrated for each river basin, including transitional and coastal waters. The "good status" in Europe's coastal and transitional waters is defined by biological quality elements, such as fish, benthic, invertebrates, and aquatic flora. Hydromorphological, physicochemical and chemical quality elements play only supporting roles (Directive 2000/60/EC).

This complex legislation required a common implementation strategy to assist states and authorities in the implementation process. Thirty-six official guidance documents and 23 technical and thematic reports have been published with the intention to provide a joint methodological approach. In 2014, a separate document dealing with the integration of ES assessments into the WFD and the Flood Directive has been added (COWI 2014). Background was observed deficiencies in the WFD implementation, especially that the benefits of a good ecological status were not included in the decision-making process and were not obvious for parts of the society.

The idea was that as soon as ecosystem functions are translated into services, a link to society is established. In this respect, ES assessments were meant to improve the holistic understanding of the environment and to visualize the societal benefits and possibilities resulting from a WFD implementation. COWI (2014) also addressed the questions where and how in the implementation process ES assessments may be applied. However, the document remained conceptual and theoretical with a strong focus on monetary valuation.

The potential benefits of ES assessments in WFD implementation are addressed in several publications (Bastian et al. 2012; Maes et al. 2012; Martin-Ortega 2012; Reyjol et al. 2014). Especially for river basins, a number of interesting and concrete approaches exist. Grizzetti et al. (2016) explored how ES concepts are used in water management, especially in WFD river basin management plans and quantify selected ES of rivers, lakes and coastal waters in Europe (Grizzetti et al. 2019). Vlachopoulou et al. (2014) and Giakoumis and Voulvoulis (2018) developed approaches that link ES and water management objectives. Vidal-Abarca et al. (2016) tested biological and hydromorphological indices used in the WFD for assessing ES in Spanish river basins. Pinto et al. (2016) carried out a contingent valuation survey in river basins to estimate the non-market benefits of ES resulting from WFD implementation. Gerner et al. (2018) demonstrated the suitability of a structured ES analysis for assessing the impact of the restoration of a German river basin on the provision, use, and benefit of ES. Roebeling et al. (2016) related the ecological and chemical status of freshwater systems to the provision of cultural ES. However, the results of Terrado et al. (2016) indicated that management

[5] http://ec.europa.eu/environment/water/water-framework/info/intro_en.htm

measures for improving the ecosystem state in river basins not necessarily improved human well-being through changes in the provision of ES.

However, Heink et al. (2016) state that "Although the concept of ecosystem services has thrived over the last ten years, its operationalization is still in its infancy." The examples further show that these studies are focused on river basins. Comparable approaches and suitable assessment tools for coastal waters, meeting a concrete WFD demand, are largely lacking.

To support the WFD implementation, we carried out an ES assessment in several lagoons, bays and in two contrasting estuaries in the German Baltic Sea region, the rural Schlei and the urban/industrialized Warnow Estuary (Schernewski et al. 2019). For this purpose, we applied a modified version of the Marine Ecosystem Service Assessment Tool (MESAT, see Chap. 25). MESAT utilizes spatial definitions, reference conditions and the good status according to the WFD as well as data and information gained during the WFD implementation process (Inácio et al. 2018). This data-based tool allowed comparative analyses between different ecological states of an ecosystem (estuary) and an evaluation of relative changes in ES provision in time (Inácio et al. 2018). Applied within the WFD context, these data-based assessments showed how the ES provision had changed between the historic, pre-industrial state around 1880 (WFD reference conditions with high ecological status), the situation around 1960 (good ecological status according to WFD), and today. Further the analysis separated the estuaries into water bodies, the smallest management units of the WFD. These case studies did show how an ES assessment can be built upon WFD, utilize the data, and visualize changes in time and between ecosystems. However, the practical value of the ES assessment for management and the evaluation of measures was limited and exemplary.

As a consequence, we applied a complementary expert-based ES assessment, where we compared the situation today with a future scenario "Warnow 2040" assuming a good ecological status as consequence of a successful WFD implementation (Schernewski et al. 2019). The assessment involved 14 scientists with different background as well as 6 experts from different regional authorities, which are responsible or at least familiar with WFD implementation. The assessments were carried out within 4 meetings, face-to-face and via teleconferences (Fig. 29.1). Our results confirm earlier assumptions (e.g., COWI 2014 or EC 2019), that ES assessments can be regarded as suitable to support public relation activities, to increase the acceptance of WFD measures and to justify the costs of the measures' implementation. Further, ES assessments are promising tools in participation and stakeholder processes within the selection and planning of measures.

29.4 European Marine Directive

The EU Marine Strategy Framework Directive (MSFD) or Marine Directive, adopted in 2008, aims to protect the marine environment across Europe. It wants to achieve a "Good Environmental Status" (GES) of marine waters and to protect the resource base upon which marine-related economic and social activities depend. The

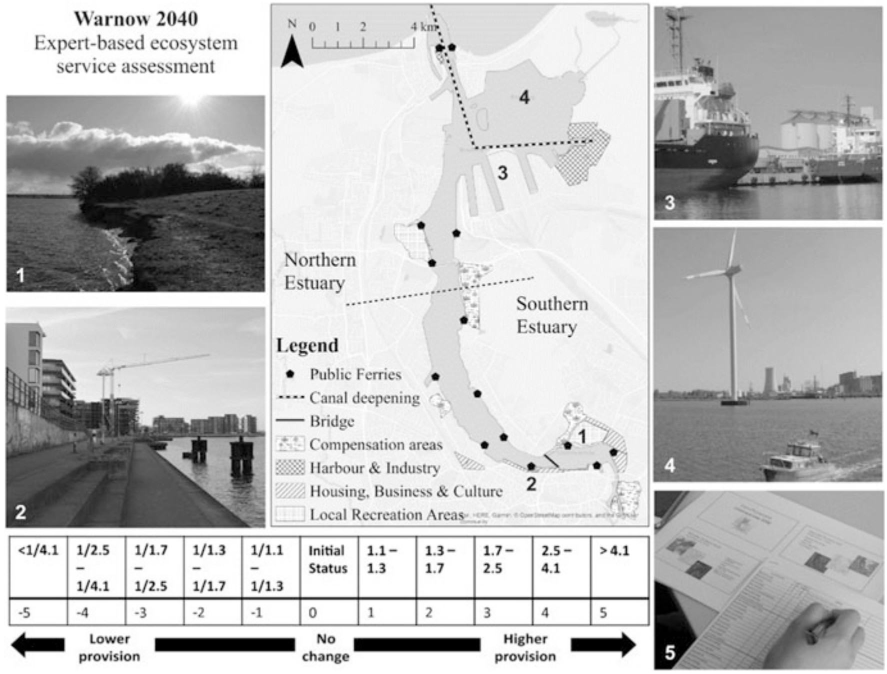

Fig. 29.1 A future vision of the urban Warnow Estuary in northern Germany (Warnow 2040). The ecosystem service provision of future vision for the year 2040 was compared to the years 1960 and 2020. Aim was to visualize the ecological changes and their consequences for the human use of the system. 2040 assumed a successful implementation of the Water Framework Directive and a good ecological status in the estuary

MSFD includes the following implementation steps[6]: the initial assessment of the current environmental status of national marine waters and the environmental impact and socio-economic analysis of human activities in these waters; the determination of what GES means for national marine waters; the establishment of environmental targets and associated indicators to achieve the GES; the establishment of a monitoring program; the development of a program of measures designed to achieve or maintain the GES. The entire implementation process is iterative and cyclical. The Marine Directive integrates basic ideas of the WFD and expands it towards the open sea. It complements the aim of a good ecological status towards a more comprehensive good environmental status. The environmental status is assessed based on 11 descriptors: biodiversity; non-indigenous species, the population of commercial fish species; food webs; eutrophication; sea floor integrity; hydrographical conditions; contaminants in the marine environment and in seafood; marine litter and underwater noise.

[6] http://ec.europa.eu/environment/marine/eu-coast-and-marine-policy/marine-strategy-framework-directive/index_en.htm

Within MSFD main documents, ES are not explicitly mentioned but it is stated that: “Marine strategies shall apply an ecosystem-based approach to the management of human activities, ensuring that the collective pressure of such activities is kept within levels compatible with the achievement of good environmental status and that the capacity of marine ecosystems to respond to human-induced changes is not compromised, while enabling the sustainable use of marine goods and services by present and future generations” (MSFD—2008/56/EC).

Borja et al. (2013) state that there is a need within coastal and marine policy, i.e., the MFSD, to protect and conserve nature, but likewise to deliver ES and societal benefits. Böhnke-Henrichs et al. (2013) deliver an ecological coastal water typology and indicators for integrating ES in marine spatial planning and management, i.e., within MSFD processes, where it can prove the link between ecological and socio-economic analysis. O’Higgins and Gilbert (2014) show how the ES concept can be embedded into the MSFD using the example of eutrophication in the North Sea. They point out that the ecosystem-based approach to management emphasizes the human dimensions of environmental problems and that incorporating the ES concept can be beneficial for reaching the MSFD objectives. Broszeit et al. (2017) show the applicability of ES and biodiversity indicators in supporting the implementation of MSFD, e.g., within monitoring efforts and reaching the GES. Caro et al. (2018) consider the assessment of ES as a crucial element to guarantee a sustainable management of marine ecosystems, as required by the MSFD. By mapping ES, the information flow between researchers and practitioners, and thus the marine management can be improved (Caro et al. 2018).

Mappings and assessments of marine ecosystems and their services within the MSFD implementation are still limited and face major challenges compared to terrestrial ecosystems. Examples are specific characteristics of marine systems, such as three-dimensionality and high temporal dynamics and the lack of data. However, ES assessments may be useful to identify needs for intervention or the regulation of human activities and offer possibilities within single MSFD descriptors.

29.5 Integrated Coastal Zone Management (ICZM)

Based on the policy document “Integrated Coastal Zone Management: A Strategy for Europe” (COM/2000/547), the “Recommendation on Integrated Coastal Zone Management (ICZM)” (2002/413/EC) has been developed and adopted in 2002. It was an important step for ICZM in the European Union, because it defined principles for a successful coastal zone management and defined tasks for EU Member States. The principles are: (1) a broad overall perspective which will take into account the interdependence and disparity of natural systems and human activities with an impact on coastal areas; (2) a long-term perspective which will take into account the precautionary principle and the needs of present and future generations; (3) adaptive management; (4) response to local specificity to their practical needs with specific solutions and flexible measures; (5) working with natural processes and respecting the carrying capacity of ecosystems; (6) involving all the parties concerned in the management process; (7) support and involvement of relevant administrative bodies at national, regional and local level; (8) use of a combination of instruments designed to facilitate coherence between sectoral policy objectives and coherence between planning and management (2002/413/EC).

After 2002, the resistance to adopt a legally binding directive on ICZM in several European Union member states increased. As consequence, ICZM lost dynamic and in 2013 merely became part of the maritime spatial planning directive (2014/89/EU). The last effort to re-vitalize ICZM at least in the Mediterranean Sea region was in 2010, when the EU ratified the Protocol on Integrated Coastal Zone Management to the Barcelona Convention (2010/631/EU). ES assessment is not explicitly mentioned in any of the documents. The influential Millennium Ecosystem Assessment, published in 2005, introduced a new framework for analyzing social–ecological systems and made ES assessments popular. However, this development was parallel to ICZM policies development in the EU and came too late to introduce ES assessment approaches in ICZM policy documents.

Olsen et al. (1997) defines ICZM as a cycle consisting of an issue identification, program preparation, formal adoption and funding, implementation and evaluation phase. The Systems Approach Framework (SAF) for an integrated assessment of coastal systems (Hopkins et al. 2011) further develops and refines the ICZM cycle (Fig. 29.2). Core is the Ecological-Social-Economic-Assessment that guides a user from the identification of an issue or problem, towards the implementation of a sustainable solution and the following monitoring and evaluation.

Many examples indicate that an ES assessment can play an important role in ICZM. This is true for concrete cases studies (e.g., Granek et al. 2010; Luisetti et al. 2011; Liquete et al. 2013b) and for estimating the net benefits of ICZM on a European scale (Williams et al. 2006). Arkema et al. (2015) conclude that embedding ES in coastal planning leads to better outcomes for people and nature. Today, several approaches exist, how ES assessments can support ICZM in practice (Arkema et al. 2015; Schernewski et al. 2018; O'Higgins et al. 2019). Baker et al. (2013) state that it requires a pragmatic, context-specific consideration how to integrate ES assessments successfully, but that ES assessments have the potential to serve as a common language for ecosystem-based management, which is the core idea of ICZM. O'Higgins et al. (2019) point out the benefits of an ES assessment, namely that it, if applied spatially resolved, supports the understanding of spatial interrelationships and connections and helps to identify key actors as well as trade-offs and synergies of different management options and related ES supply.

We applied an ES assessment to compare different alternative coastal water management measures. Based on this analysis, we critically evaluate the potential role of ES assessments in public participation and in the context of the System Approach Framework and ICZM (Schernewski et al. 2018). Chapters 24 and 25 provide complementary case studies. In general, our expert-based approaches show that an ES assessment can help to catch the views of experts and stakeholders, extract disagreements between opinions, reveal misunderstandings and misperceptions and

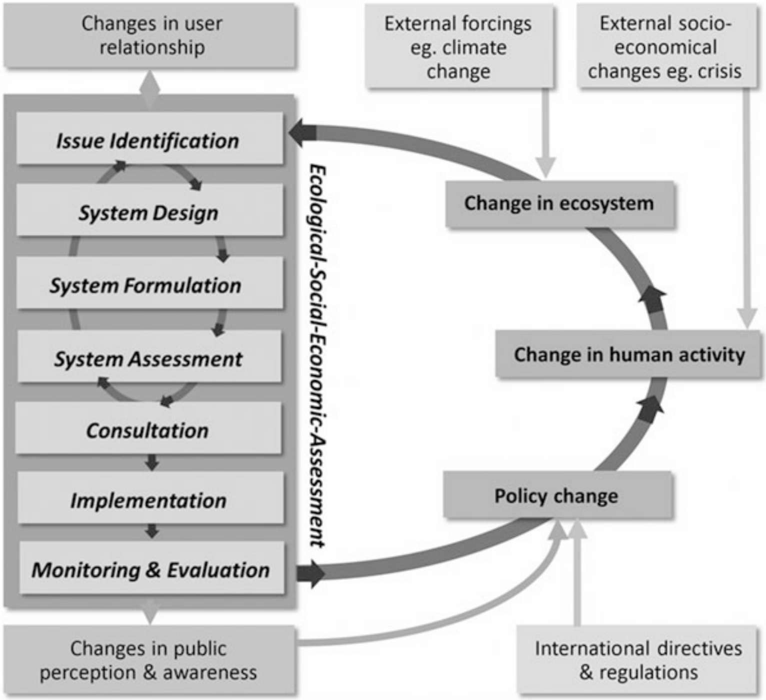

Fig. 29.2 The Systems Approach Framework (SAF) for an integrated assessment of coastal systems modified after Hopkins et al. (2011)

allows to define services of highest priority for the local community. The results can be used for guiding expert and stakeholder discussions and for harmonizing views. ES assessments can support ICZM in different respects and different states of the process. EC (2019) points out that ICZM in combination with strategic environmental assessments (SEA's) may serve as good framework for integrating ES into coastal planning.

29.6 Maritime Spatial Planning Directive

Competition for maritime space—for renewable energy plants, aquaculture, and other uses—has highlighted the need for managing our waters more coherently. Maritime spatial planning (MSP) works across borders and sectors to ensure human activities at sea take place in a more efficient, safe, and sustainable way. That is why the European Parliament and the Council have adopted "legislation to create a common framework for maritime spatial planning in Europe" (MSP-Directive).[7]

[7] https://ec.europa.eu/maritimeaffairs/policy/maritime_spatial_planning_en

Maritime spatial planning means a process by which the relevant EU Member States' authorities analyze and organize human activities in marine areas to achieve ecological, economic, and social objectives.[8] Outside of Europe, it is often referred to as marine instead of maritime spatial planning. The term "marine" reduces the emphasis on development and stresses nature and environmental protection.

MSP is a spatial and holistic process promoting coherence with environmental and sectoral policies. This includes to achieve a Good Environmental Status of marine ecosystems (see MSFD), a Good Ecological Status of coastal and transitional waters (see WFD), favorable conservation status of habitats and species (see Biodiversity Strategy and Habitats Directive), and the Action Plans of the European Regional Sea Conventions.

Benefits of MSP are to: reduce conflicts between sectors and create synergies between different activities; encourage investment—by creating predictability, transparency and clearer rules; increase cross-border cooperation—between EU countries to develop energy grids, shipping lanes, pipelines, submarine cables and other activities, but also to develop coherent networks of protected areas; protect the environment—through early identification of impacts and opportunities for multiple use of space.

The MSP Directive lists several minimum requirements that should be taken into account in maritime spatial plans, such as: land-sea interactions; the ecosystem-based approach to management; coherence between MSP and other processes such as integrated coastal management; the involvement of stakeholders; the use of best available data; transboundary cooperation between Member States; and cooperation with third countries.

In Europe, the 23 coastal Member States are obliged to develop a national maritime spatial plan by 2021. They are free to design and determine the format and content of their maritime spatial plans, including the institutional arrangements and the allocation of maritime activities. The online European MSP Platform provides an insight into the progress.

The ES concept can be used in different steps of the MSP process (Schernewski et al. 2018), which follows a continuous, adaptive, and ecosystem-based approach. When defining the MSP area, the ES concept can support in identifying all ongoing activities therein, as they are based on the marine ecosystems and its services. Stakeholders can be involved in order to identify and list ES. At the policy formulation level, ES assessments can be used to define concrete objectives that could safeguard and enhance marine ecosystems and their conservation. Mapping marine ES can also support identifying critical areas for intervention and facilitation of managing maritime activities.

Several studies confirm the increasing importance of ES in coastal and marine planning and its outcomes (Arkema et al. 2015; Guerry et al. 2012) and propose analytical frameworks, tools or analyses for MSP (Ivarsson et al. 2017; Lester et al. 2013; White et al. 2012). Some case studies assess ES within an MSP context

[8] https://eur-lex.europa.eu/legal-content/EN/TXT/HTML/?uri=CELEX:32014L0089&from=EN

(Domínguez-Tejo et al. 2016; Nahuelhual et al. 2017; Outeiro et al. 2015) and single EU Member States, such as Latvia, successfully assessed marine biodiversity and ES within the MSP process.

Guerry et al. (2012) show how the marine modeling tool of the "Integrated Valuation of Ecosystem Services and Tradeoffs" (InVEST) including also monetary valuation can be used for stakeholder involvement and as a decision-support tool within coastal and marine spatial planning. As monetary and biophysical assessments dominate spatial planning data, Klain and Chan (2012) state that there is a lack of cultural valuations showing results of participatory mapping with focus on cultural ES and Rodrigues et al. (2017) provide an overview about the state-of-art. Böhnke-Henrichs et al. (2013) developed a typology and defined indicators for the assessment of ES for marine spatial planning and management. Based on this, Von Thenen et al. (2019) present an indicator pool for expert-based ES assessments that can be used to structure processes and to analyze future scenarios within MSP.

Similar to the MSFD, challenges when integrating ES into MSP are the three-dimensional character of water bodies and the high spatio-temporal dynamic of core parameters, such as temperature or salinity (EC 2019). Further, the lack of understanding how ecological functions and processes effect some marine ES remains a shortage. MSP is a conceptual framework approach. Therefore, the application of ES assessments will mainly remain on a general, informal level and may support public participation.

29.7 Other Coastal and Marine Policies

The EU Floods Directive (Directive 2007/60/EC), entering into force in 2007, requires EU Member States to assess the risks of flooding of their water courses and coast lines. This includes the mapping of flood extent, assets of the infrastructure and humans at risk in these areas as well as the implementation of adequate measures to reduce this flood risk.[9] Therein, the term ecosystem services is explicitly mentioned in two communications of the European Commission on water policy, one published in 2012 as "A Blueprint to Safeguard Europe's Water Resources" (COM/2012/0673), and another one in 2015 namely "The Water Framework Directive and the Floods Directive Actions towards the 'good status' of EU water and to reduce flood risks" (COM/2015/0120). Flood regulation usually is considered as a regulating ES in most ES frameworks, which establishes a natural link between ES and the Floods Directive. Liquete et al. (2013a, b) consider the assessment of the regulating ES "coastal protection" as supporting for the implementation of the EU Floods Directive. The authors see potential specifically in supporting the national coastal flood hazard and risk maps of each Member State. ES can also be integrated within the flood risk management plan development, addressing the obligations of

[9] https://ec.europa.eu/environment/water/flood_risk/

the Floods Directive (Grizzetti et al. 2016). Another opportunity for ES assessments is public involvement and information in the planning process.

The EU strategy on adaptation to climate change, adopted in 2013, aims to make Europe more climate-resilient by enhancing the preparedness and capacity of all governance levels to respond to the impacts of climate change.[10] It explicitly mentions the ES concept in its supporting documents. The European Commission (EC 2019) states that the ES approach could play a supporting role within implementation processes of the EU long-term strategy and the EU Climate Adaptation Strategy. ES assessment results could be used for promoting nature-based solutions, showing benefits of potential carbon sequestration projects (e.g., restoration of peat bogs), other climate adaptation measures to increase the resilience of ecosystems, or the protection of coastal areas from flooding and storms (e.g., marine seagrass meadows). A potential use of ES assessments and its quantification is shown in Dunford et al. (2015) providing an overview of potential future impacts on ES by climate and socio-economic changes. Chapter 26 provides another example.

The United Nations´ 2030 Agenda for Sustainable Development aims to eradicate poverty and achieve sustainable development by 2030. Goal 14 aims to conserve and sustainably use the oceans, seas, and marine resources for sustainable development. Considering the three dimensions of sustainable development (economic, environmental and social), the ES concept bears potential to provide an overarching framework to this in a structured way by integrating and combining different ecosystems and their related socio-economic systems (Bouwma et al. 2018). Wood et al. (2018) believe that ES can help to achieve SDGs, as they are often underpinned by the provision of at least one ES as the benefit of nature to humans. The European Union is strongly committed to the Agenda 2030 and in 2017 signed a new European consensus on development "Our world, our dignity, our future" as a guiding document for policy.

The EU Blue Growth Strategy supports sustainable growth in the marine and maritime sectors as a whole, as seas and oceans are drivers for the European economy having great potential for innovation and growth. It is the maritime contribution to achieving the goals of the Europe 2020 strategy for smart, sustainable and inclusive growth.[11] Several official documents mention the ecosystem approach and services from the marine ecosystems explicitly. Blue Growth shall be under safeguarding biodiversity and protecting the marine environment, thus preserving the services that healthy and resilient marine and coastal ecosystems provide.[12] Further one Agenda 2030 goals is to conserve and use the oceans, seas and marine resources sustainably. Therefore, marine resources shall be used sustainably, enable healthy marine ecosystems and a strong blue economy.[13] Lillebø et al. (2017) show

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[10] https://ec.europa.eu/clima/policies/adaptation/what_en#tab-0-0

[11] https://ec.europa.eu/maritimeaffairs/policy/blue_growth

[12] COM/2012/0494. Blue Growth opportunities for marine and maritime sustainable growth: https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX:52012DC0494

[13] https://ec.europa.eu/maritimeaffairs/sites/maritimeaffairs/files/swd-2017-128_en.pdf

how marine ES can support the Blue Growth agenda that requires coordination of trade-offs between economic, social and environmental aspects, for example, by showing different sectoral interests and balance them.

Other coastal and marine policies where ES may play a supporting role are the Bathing Water Directive, Fisheries Policy, Regional Sea Conventions (RSCs), Baltic Sea Action Plan (BSAP), or Environmental Impact Assessments (EIA).

29.8 Synthesis

In 2019, the European Commission published the working document called “EU guidance on integrating ecosystem and their services into decision-making” (EC 2019). Based on this document, by our case studies and experiences from southern Baltic coastal systems (Inácio et al. 2018, 2019; Karstens et al. 2019; Robbe et al. 2018; Schernewski et al. 2018, 2019), we provide a general, summarizing overview where and when ES assessments are useful in the implementation process of different policies (Fig. 29.3).

Taking into account the European Commission expectations (EC 2019) and our experiences, we can further conclude that ES assessments have the benefits to

- Establish a link between humans and nature and introduce a socio-economic view on nature protection, preservation, and sustainable use.
- Support communication with and information to the public and offer new possibilities for a mobilization and a guided, active involvement of stakeholders.

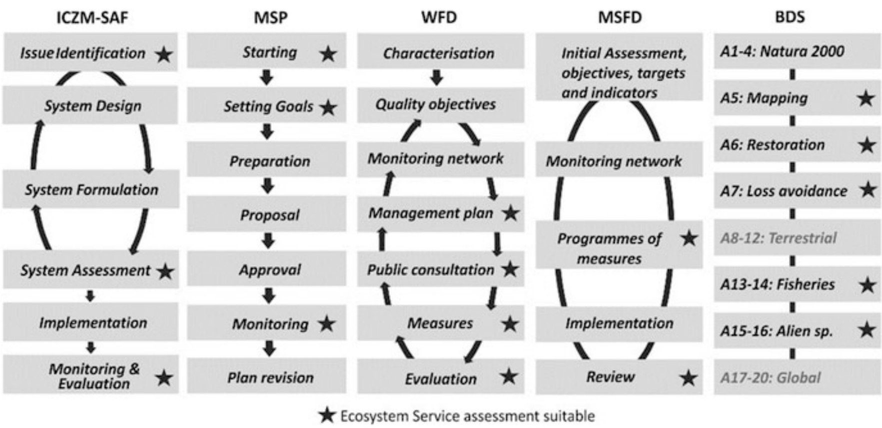

Fig. 29.3 Options for applying ES assessments during the implementation of different policies: ICZM-SAF (Integrated Coastal Zone Management); MSP (Maritime Spatial Planning); WFD (Water Framework Directive); MSFD (Marine Strategy Framework Directive) and the defined 20 actions for implementing BDS (Biodiversity Strategy). Expanded after Schernewski et al. 2018)

- Allow the comparison of different ecosystem states in the past, presence, and future and across regions and a more comprehensive definition of targets and policy objectives.
- Indicate and quantify conflicting interests in and between services and.
- Enable the comparison and prioritization of different environmental measures, conservation and restoration approaches, or management concepts.

Originally, one of the ideas of ES assessments was to provide additional arguments for nature conservation (e.g., a monetary view on the benefits humans obtain from nature), which could be used to justify costs for nature protection and increase the public acceptance of expenditures (EC 2019). This might work and might be reasonable in defined, concrete cases. However, because of all methodological problems and unsolved challenges linked to monetary ES approaches, we doubt that this will be beneficial for coastal and marine policy implementation at a large and general scale. It simply seems not possible to assess economic values of all ecosystem services in a reasonable timespan. An economic valuation may translate the value of nature into a generally understandable monetary dimension. But this approach and view bears risks. It might question the largely accepted view that nature has an intrinsic value, a value in itself. As soon as we express values monetary, we raise the expectation that if the values provided by nature are not high enough, we may choose an ongoing exploitation. High monetary values of cultural services might increase tourism, intensify pressure and favor exploitation instead of protection. This is why ES assessments should always include socio-ecological-economic aspects.

Bull et al. (2016) provided a first general Strengths-Weaknesses-Opportunities-Threats (SWOT) analysis for ES frameworks. This serves as basis for an own summarizing SWOT analysis focusing on ecosystem service assessments supporting European coastal and marine policy implementation (Fig. 29.4). Altogether, we can conclude that ecosystem service assessments are high on the political agenda and raise many expectations, which hardly can be fulfilled.

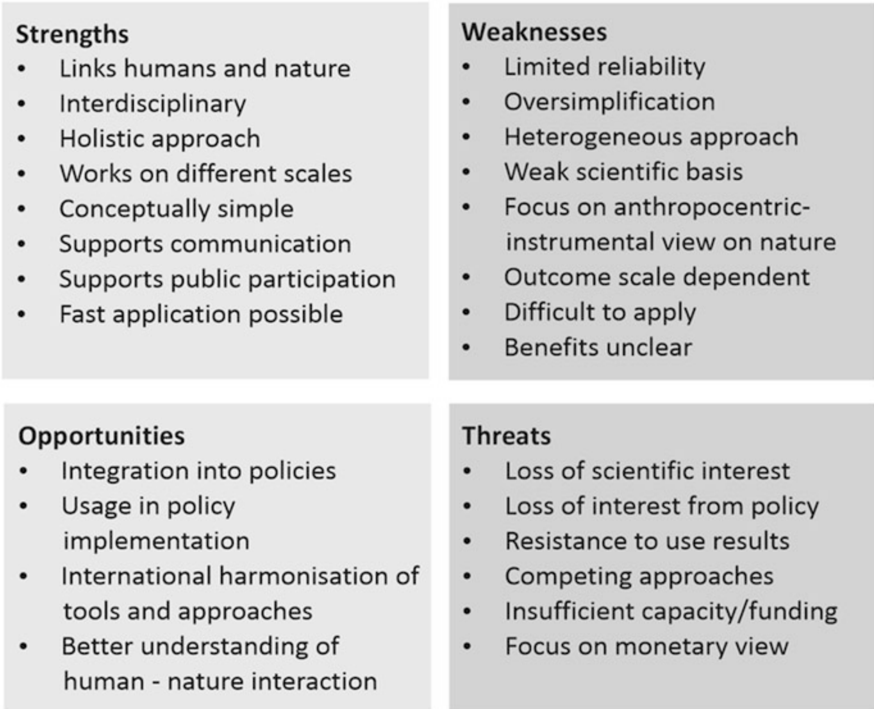

Fig. 29.4 SWOT Analysis of ES assessments in European coastal and marine policy (modified after Bull et al. 2016)

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Ecosystem Services and Sustainable Development: The Case for Strong Sustainability

30

Konrad Ott

Abstract

This Section draws some conclusion with respect to the underlying concept of sustainability. It is argued that the overall approach in this volume and its many empirical findings and policy suggestions imply the general concept of strong sustainability. From the background of the history of sustainability and with reference to recent literature on marine sustainability, the argument in favor of strong sustainability can be substantiated. Finally, the building blocks of this volume constitute a robust theoretical approach.

The overall argumentation in this volume and the many empirical findings imply policies of sustainability with respect to ecosystem services. In this final outlook, we wish to reflect upon underlying concepts of sustainability, which can ground generic rules and obligations to preserve and restore at different spatial scales the stocks and funds of natural capital from which ecosystem services flow. The cascade-model explains the nature-stock-flow-benefit-values-connectivities well, but it cannot substantiate normative obligations.

The idea of sustainability has a long tradition. The word "Nachhalt" was coined in 1713 within a book on forestry, written by von Carlowitz (1713). According to Carlowitz (1713), harvesting timber is to be allowed only within the limits of the natural growth-rates of trees. Such mandatory constraint has been justified with respect to the legitimate interests of future generations. From the traditional perspective, sustainability is an ethical idea for the economics of natural resources. Thus, sustainability was seen as a conceptual constraint over the long-term use of natural

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resources, seen as “living funds” (in current terminology). In the 19. century, this idea was applied to fertile soils by Justus von Liebig and to marine resources by Karl August Möbius (Ott 2021 with further references). The idea of sustainability, however, was lost within the decades of warfare, revolution, civil wars, and genocides between 1914 and 1945. It was marginalized in the post-war decades of GDP-growth, consumerism, and welfare states. Its renaissance originated in the environmental movements in the 1960ies. Recognizing limits to growth (Meadows 1972) implied the inconvenient message that humanity had to cope with natural limits (today: planetary boundaries), the transgression of which could turn out to be catastrophic. Many bleak prediction of this report did not realize so far, but the sense of alarm should be kept alive in the epoch of the Anthropocene.

In 1987, UN-commission (World Commission on Environment and Development, WCED) launched its report “Our Common Future” (WCED 1987). The WCED adopted the mission to reconcile growing environmental concerns (pollution, extinction of species, desertification etc.) with the ideas of progress, development, wealth, and growth in a postcolonial age and within a “cold war” between competing super-powers. Most members of this commission were deeply divided ideologically on almost all matters of substance. Finally, the WCED reached a common moral denominator: basic needs of poor humans should be fulfilled. From this “basic-needs”-approach the most famous recent definition of sustainability was coined. The long-term constraint did not refer to natural funds any more, but to the ability of future generations to fulfill their own (basic) needs and (some) aspirations. Under this “(basic)-needs”-constraint, the WCED favored an even more intense utilization of natural resources. It seems fair to say that WCED shifted the idea of sustainability from a more conservationist paradigm to a more humanitarian and social justice paradigm. This definition became rapidly prominent within UN-circles and it was adopted at the Rio summit in 1992. The Sustainable Development Goals (SDG’s) are clearly in line with the WCED’s definition.

We shall not present a persuasive case in favor of the “best” general concept of sustainability (see Neumayr 2013). It seems, however, safe to argue that at least three Sustainable Development Goals (SDG) that directly refer to climate, biodiversity, and ocean implicitly presuppose the concept of strong sustainability (see Neumann et al. 2017 for SDG 14). Visbeck et al. (2014) outline the scope of research as being entailed in SDG 14. Franke et al. (2020) propose to interpret the metaphor of “ocean health” which was coined by SDG 14 in terms of resilience, productivity, and diversity. The problem of ocean acidification has been addressed by Böhm and Ott (2019) from an environmental ethics perspective. This monography reflects upon lines of reasoning in environmental ethics and applies them to the long-term problem of acidification, focusing endangered coral reefs. Neumann et al. (2017) apply the concept of strong sustainability specifically to sub-target SDG 14.5, which demands to protect at least 10% of the planet’s coastlines. These lines of reasoning in conjunction with crucial findings on marine ecosystem services in this volume seem sufficient to adopt the concept of strong sustainability (Daly 1996) in marine environmental affairs.

This concept is structured into the layers of (a) ethical grounding, (b) principle, (c) rules and guidelines, (d) special fields of interest, (e) objectives, sub-targets, and indicators, and (f) special models for implementation (Ott and Döring 2011). On the supreme layer of ethical grounding, strong sustainability would adopt two sources of normativity, as environmental ethics (Ott 2020) and justice (Ott 2014), giving special attention to environmental legacies within a chain of generations. It also proposes an integrated model of environmental evaluation (Ott et al. 2021, see also Chaps. 5.3, 5.4). Strong sustainability is open for non-anthropocentric concepts of inherent moral values in nature). It might ban whaling, while allow fisheries and sustainable aquacultures (for aquacultures, see Ott et al. 2020).

Strong sustainability can adopt the original principle of Aldo Leopold's "land ethics" as it wishes to sustain the fertility/productivity, resilience, and diversity/richness of terrestrial and marine systems for the sake of future generations (and, perhaps, for the flourishing of sentient beings within their habitats) (Neumann et al. 2017; Franke et al. 2020). The following wording of the principle might be appropriate: "*Use all land and sea only in such ways and kinds which preserve or enhance the fertility, resilience, and diversity of biotic and ecological systems. Don't act otherwise. If fertility, resilience, and diversity have been impaired in the past, try to restore them fully, if possible.*" Neumann et al. (2017), Böhm and Ott (2019) as well as Franke et al. (2020) define the ocean as "healthy" if and only if it is resilient, productive, and diverse. Productivity refers to providing services, resilience covers regulating services, while diversity and richness point to cultural services. Violations of this principle impair the flow of service. Any principle can be overridden but it also sets the bars high for the burden of proof.

Any interference with nature underlies this *prima-facie*-principle. Even if one might concede that pressing humanitarian objectives may override this principle, such concession would hardly hold in the wealthy countries surrounding the Baltic Sea. If so, the principle clearly holds with respect to marine systems of the German Baltic coastlines. These coastlines should become a paradigm case for actualizing SDG 14. 5 on coastal protection. This general principle can be specified to a set of rules. Since strong sustainability casts doubts on the economic hope that natural capitals might be substitutable by man-made or human capital, it adopts the basic rule to hold natural capital (at least) constant over time (Daly 1996; Neumayr 2013; SRU 2002; Ott and Döring 2011). This constant natural capital rule is specified to a set of management rules. This set includes a rule to invest in natural capitals (to enhance fertility, resilience, diversity) if such capitals have been depleted in the past. This investment rule demands recovery and restoration of degraded ecosystems, protection of viable populations of species, and even strictly protected areas. As a methodical measure, strong sustainability focusses stocks and funds of natural capitals as they change over time for better or worse (Klauer et al. 2017). This measure also applies to ecosystem services. The rules and measures are reflected in the guidelines of (a) consistency of economics within boundaries, (b) enhanced resilience of land-use systems, and (c) (more) sufficiency in cultural lifestyles and patterns of consumption, including tourism.

The long argument being presented in the previous chapters of this volume implies the conclusion that one cannot adopt the crucial findings of ecological research, economic assessment, and ethical reflection on ecosystem services *and* reject the concept of strong sustainability. If one adopts a bundle of premises A which imply (either semantically or pragmatically) B one cannot, by pain of inconsistency, adopt a partial or complete negation of B.

In German constitutional law, the natural preconditions of human life have to be protected in the interest of future generation (Art. 20a GG). In German politics, a national sustainability strategy which comprises biodiversity and adaptation strategies has been adopted twenty years ago. Such integrated strategies should be mobilized in a post-Covid-situation, seen as window of opportunity for transition (Popp and Ott 2020).

The lines of reasoning in environmental ethics, the method of ecosystem service assessment, a Leopoldinian interpretation of "ocean health," the concept of strong sustainability, SDG 14 and its sub-targets, the case studies on the Southern Baltic Coastal Systems, the directives and policies of EU and, last but not least, German constitutional law are coherent normative, methodical, and empirical building blocks of a presumptive robust interdisciplinary theoretical concept in marine sustainability science. Critics may consider strategies of falsification.

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31 Concluding Remarks

Felix Müller and Hendrik Schubert

Abstract

This last Chapter includes some summarizing conclusions about the contents of the papers in this book. After sketching the specific characters within the sequence of articles, the role of environmental valuation and ecosystem services is discussed, and finally some answers and comments related to the ten initial questions asked in Chap. 1 are formulated.

The above texts, figures, maps and tables have unfolded various attempts for a **comprehensive synoptical** analysis of the coastal ecosystems of the southern Baltic. Several **spatial ranges** and **scales** have been covered (hinterland to offshore), aspects of ecological **structures and functions** (matter cycling, interactions between elements as well as subsystems, limitation patterns), **societal perception** and human **demands** (from economics via cultural to ethical aspects) have been displayed and integrated. The coastal conditions have been observed by **linkages of spatial and temporal perspectives,** and several **disciplinary achievements** resulted in a sound knowledge base for compiling an overview about the ecosystem service patterns of the recent coastal systems, embedded by in-depth knowledge not only about the Baltic system in focus, but also the **bordering hinterland and offshore systems.** Especially this aspect of an integrative assessment of the coastal region as a transition area between terrestrial and offshore ecosystems was both, a challenge

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as well as a source of new insights into coupling of matter fluxes and feedback mechanisms. Investigating a large spectrum of geomorphological coastal types, ranging from cliff coasts to lagoonal systems allowed for refined analysis of ecological, cultural and socioeconomic effects of anthropogenic impacts on the hinterland as well as the offshore ecosystems with the coastal stripe being impacted by both of them. The lack of regular tides in the study area, restricting water exchange with the adjacent Sea for lagoon systems and estuaries, allowed for high spatial resolution of matter fluxes in transition ecosystems and, consequently, refined assessments of terrestrial impact pathways to the Baltic Sea. These results, outlined in detail in Chaps. 9–18, were required to create a knowledge basis uniform with respect to spatial as well as temporal resolution.

Realizing the target function of better understanding **ecosystem services (ES),** the **interdisciplinary approach** was promising, but it also revealed several problems and **uncertainties** when tested—namely visible divergences in assessing individual ES. Surprisingly, these differences were not (exclusively) due to different disciplinary backgrounds, starting points and receptions. Much more, part of the problem was due to **uncertainties in the definition of ES** and could be solved by sharpening them—a good mutual description what the individual ES are comprising of was found to be crucial. Another part of variability was caused by **different depths of specific knowledge**, especially when estimating the consequences of change. Discussions of the respective feedback mechanisms and interactions led to a reduced variability and, at the same time, to an intensive exchange of knowledge, allowing better understanding of the mechanisms behind also for colleagues from other disciplines. Another basic obstacle arises from the implicit **anthropocentric approach** of the ES concept. It seems to be located in a strict contraposition to the basic protection motivation of several colleagues who favour the **inherent values of nature**, giving nature a high value in itself, not as a service provider. From such a viewpoint, **monetary valuation** cannot be accepted, while other pathways of arguments stress economic values as the focal linkage between human and environmental systems (see Chaps. 5.2 and 29). Besides these points, ES-related problems were obvious concerning

- Their general complexity, wingspan and diversity
- Spatiotemporal and functional scale mismatches
- Data scarcity, data handling and linkage with monitoring activities
- Very different traditions and languages of different disciplines
- A variety of inherent targets of the single disciplines and sometimes missing mutual respect and patience
- Openness for linking qualitative and quantitative approaches
- And resulting “human problems”

A main achievement, improving the instrument significantly, was the **definition of a reference scenario** for a robust **assessment of ES**. This should not be seen as an obstacle or additional burden, but an integral part for applicability. It is sharpening the analysis for a specific project, avoiding comparisons with irrelevant scenarios and thus preventing useless debates.

The ES instrument has shown its capacity to raise awareness for sustainable management, to illustrate pressures and historical changes as well as consequences of management scenarios and to provide a platform for solution-focused discussions. It can be utilized in preparing balanced solutions and to derive a focus for **research demands**. With respect to the latter point, the application approaches performed identified multiple **knowledge gaps** which should be filled in by forthcoming research in order to increase robustness and reliability. Some of them are:

- Widening the knowledge base about the mechanisms of creation of ES
- Closing consequent lines of argumentation between ecosystem integrity, condition, service and value under potential-related and flow-related viewpoints
- Concentrating on a detailed quantitative comparison of the different potentials to provide key services comparing terrestrial and marine ecosystem types, concentrating on regulating ESS capacities
- Discussing, to what extent standardization of the process of ESS quantification is required
- Finding out if uncertainties in assessing the consequences of changes in management are due to lack of knowledge or missing evidence?
- Conducting time-series of ESS patterns in well-studied ecosystems in order to test forecast potentials by means of long-term data series
- Investigating and formulating the interrelations between ecosystem services and their roles in trade-offs
- Improving the applicability of the approach and collaborating in management projects

The investigations have also shown that there is a high potential to keep on developing a linked conception of ecosystem-based issues towards an elaborated system of **sustainability assessments.** Obviously, the respective database must be founded on a suitable knowledge base concerning the environmental and human preconditions. That are long-term features on broader spatial and temporal scales (**location characteristics**, see Chaps. 3–8). They are providing the baseline status for the ecosystem **structures and processes** (see Chaps. 9–18), which should include features of abiotic heterogeneity as well as biodiversity on the structural side. A functional characterization should include the flows of C, N and P as well the water and energy budgets. These ecosystem components are linked to the concept of **ecosystem integrity**, which is reflecting the ecosystem state of the DPSIR indicator scheme (see Chaps. 19–26) as well as the ecosystem condition of the European ES concept. The **ecosystem services** (see Chap. 20 ff.) are building a bridge towards the **human subsystems**, such as economical, social, legal, technological or cultural aspects (see Chaps. 19–26).

The preceding papers thus can be understood as steps on a pathway to find such an integrated and emergent comprehension of the elements of a sustainable development. Again, we have to state that the arising strategic and practical **complexity** is overwhelming, especially if coastal systems are involved, because here we have to link very different functional and structural units of space and time. Furthermore, different pressures are modifying these systems in different causal networks, and climate change will be transferring them with different and unexpected consequences. Nevertheless, it will be inevitable to face those difficult problems in the future and continue developing instruments for the analysis and management of coupled human-environmental systems, especially for the complex coastal zones.

To round down the overall conception of this book, we will finally try to combine **some answers and comments** related to the questions asked in Chap. 1. We started the texts with a rather theoretical analysis of interdisciplinarity, illuminated from the position of science structure and science theory.

Q4: What are the demands of coastal research and management for cooperation between the involved scientific disciplines, and how has the attained interdisciplinarity been applied in this book?

With the question Q4 we asked for the demands for **interdisciplinary** cooperation in the light of the problematic environmental situation of the Baltic coastal ecosystems. The paragraphs written above are including the answers: Human activities have produced enormous and unknown pressures on the coastal ecosystems and the consequences can only be foreseen and mitigated if we continuously develop and apply an **integrated human-environmental systems approach** in order to attain a sustainable development. The concepts of ecosystem integrity, ecosystem condition and ecosystem services are focal components of that framework. If we want to realize such an approach, a basic requirement is to accept the enormous **complexity** of our coastal object, to improve our instrument to cope with the inherent complicatedness, to open our minds to more holistic approaches and to surmount the borders of scientific disciplines or institutional limitations.

Q5: Which are the basic environmental conditions of the research area of the following chapters?

The objects of that integration have been presented in Chaps. 3–8, following Q5 and summarizing the ecological situation of the **study area** referring to several layers of ecological constraints. Thereby, some special characteristics of the Southern Baltic coasts have become obvious, especially the pronounced small-scale heterogeneity of coastal geomorphology, creating a patchwork of cliff coasts, sandy beaches and different kinds of coastal inlets and, in front of them, a large variety of seafloor types. The latter two subsystems, estuaries and seafloor, received special attention because of their spatial heterogeneity.

Q6: Which are the basic ecosystem mechanisms, interrelations and patterns in the respective habitats, and which is their seasonal and long-term variability?

Q7: Can this knowledge help to provide a sound ecological data-base for human-environmental systems analysis?

The complex answers for question 5 give us an impression of the environmental background of the important ecological processes. Concentrating on the aquatic ecosystems, Chaps. 9–18 is targeting on the questions 6 to 8. Here, some basic ecosystem mechanisms, interrelations and patterns are described from structural and functional viewpoints. With respect to **nutrient and carbon dynamics**, pools, stores, flows, inputs and losses, have been quantified for selected sites, the networks of energy, information, matter and nutrients flows have been described and also direct and indirect effects of toxic substances have been investigated and described in Chaps. 9–18. Thereby, the impacts of the enormous **terrestrial inputs into the Baltic Sea** have become obvious, and their effects on food webs, living conditions and ecosystem services are getting obvious again and again in the preceding chapters.

Seasonal and long-term variabilities have been illustrated. In this context, some parameters and indicators have shown extraordinary dynamics, e.g. macrophytes. The added value of performing ecological studies on a variety of time scales becomes most obvious by the experiments about P-uptake, challenging once again the established point of view about limitation regimes in hypertrophic lagoon ecosystems (Chap. 18).

Overall, the discussed variables provide a sound ecological database for human-environmental systems analysis. They can, on the one hand, satisfactory be used to analyze the ecosystems' states and to make relevant political decisions. On the other hand, it is visible that good sectoral monitoring schemes and statistical regularities are available, but their integration does not take place. We need more **cooperation** between science and administration, and between different branches of science and management. Moreover, as becoming visible from the previous chapters, a sound database about changes in ESS distribution over time is crucial for analysis of human impact on ESS provision and intensity of use. Therefore, developing a new **human-environmental monitoring and evaluation scheme** with early-warning potentials is an important future task. The preceding papers show that adapted databases are in development as well as the linkage between human and ecological approaches. Due to administrative structures, data availability sometimes is difficult; therefore, also here cooperation and application of new concepts must be improved.

Q8: How do the investigated ecosystems react after human modifications, which is their reactivity, resilience and adaptability?

Coming back to the initial questions, Q8 generally asks for the **reactions of the investigated systems** after human modifications, demanding features of their reactivity, resilience and adaptability. In this context it has been shown that eutrophication, N flows, P accumulation, toxic inputs and pollutants, fishery, alien species, touristic pressures, shipping, technical coastal protections plants/facilities, and climate change, etc. are well-known sources of pressures. They have been partly understood and countermeasures are discussed and familiar. But the **mutual accelerations** through feedback loops and indirect effects are not well-studied at all. Here, additional activities should be demanded from science as well as management. Otherwise, the focal destructive modifications may rise: simplification by species extinctions, losses of nutrients, information, energy storages, decrease of integrity and health, reduced exergy capture capacity, increasing respiration and entropy production, reduced storage capacity, efficiency, cycling potential, reduced organismic water flows are consequences of such **retrogressive dynamics**, as it has been demonstrated by several case studies in this book. Often, the dominance of long-term nutrient loadings and the following eutrophication processes are the main triggers of such developments.

Q 8 finally asks for the **resilience of the investigated systems**. It is the implicit target of many protection actions in order to make ecosystems elastic so that they can return to the old stages, when the living conditions were "fine". We have been looking at the long-term data sets and the questions has arisen: "Is resilience an illusion?" On the one hand, all ecological reactions are irreversible processes; therefore, it is a fact that not an identical but only a similar system can be re-established after a regeneration phase. Knowing this and facing the rapid climate change dynamics, the question is coming up whether on the other hand, whether resilience really makes sense: While we restore an old-fashioned system, the exterior conditions might change enormously, so that the restituted result is no more suitable due to the changing constraints. As resilience can be seen as a "tolerant expression of stability", and as we are discussing about sustainable development (instead of sustainable stability), we might start looking at this concept also a little critical. Looking at the example of our Bodden ecosystems this meaning of resilience becomes clear also in a functional manner: the ecosystems have become resilient, maintaining not the ecological target state but regularly coming back to the disturbed situation. Consequently, it might be opportune to change the target functions from resilience to **adaptability**, allowing the systems to adapt to the prevailing conditions and to unfold their potential for complexification, thus optimizing their integrity independent of what has been at the site before. Such self-organization-based management might be risky from a species-protection viewpoint, but it might favour more healthy conditions and a higher long-term sustainability of the overall **human-environmental systems**.

Q9: Which are the focal mechanisms, inter-relations and patterns of the societal aspects in order to provide a sound knowledge base for human-environmental systems analysis?

The ninth question (Q 9) demands the human influences within the study region. In Chap. 19, these items have been discussed theoretically in a large extent, leading to a concept of **human-environmental systems**, which is based on a sequence of the ecological approaches of ecosystem integrity, ecosystem condition, ecosystem services with systems' adaptability as a key target function. By linking this ecological branch with the basic human elements of the DPSIR argumentation, it is possible to develop the idea of sustainable management targets in general. It can be accomplished by respective available methodologies. But it is linked with a modified viewpoint of adapted ecosystem dynamics, species protection and resilience. If we wish to reach an optimal ecological and anthropocentric functionality, structural losses and changes in conservative management visions have to be expected.

Q10: Which are the most effective ecosystem services in the research areas, how can they be described and indicated and how can we derive them from linked ecosystem analysis linked with societal approaches?

Moreover, the conceptual offers of the **ecosystem service** approach, which is the object of our initial question Q 10, should be accepted, applied and developed. In Chaps. 20–26, several approaches are outlined, exemplified and demonstrated, reaching from ethical observations over narrative value discussions and elaborated expert judgements or matrix applications to empirical measurements, statistical analyses and model applications. Bundles of services have been analyzed, mostly referring that a balanced ecosystem service optimization must be based on multiple services; whenever one service type alone becomes superior, the situation should be observed critically. Consequently, in spite of consciously accepted or even desired local exceptions, all services play important roles; they should always be evaluated as service groups. The papers of Chaps. 20–26 as well as the literature situation furthermore show that good opportunities are available for ES qualification, quantification and indication. We have learnt a lot about ES production, there are many interesting approaches of valuation and tools are available on many different methodological degrees of complexity.

One interesting aspect, which is still in an initial stage, is the comparison of the **ecosystem service potentials of marine, freshwater and terrestrial ecosystems**. Thus, the matrix approach presented in Chaps. 20–26, is still accompanied by some uncertainties: While we know much about the single medial ecosystem classes, we still see a knowledge gap in the quantitative comparison between them. The respective open questions are founded in our traditional and disciplinary methodologies. Marine scientists have to use other instruments than forest scientists, and consequently an overarching comparison is still a challenge. Nevertheless, the last paragraphs of Chaps. 20–26 show that the question for realistic application of the ES approach can be answered positively: The available data can be used to develop scenarios on coastal ES dynamics and to evaluate the outcome of recent and potential future conditions. Of course, the basic condition for the inherent optimism of this answer is the observer's personal significance of quantitative exactness: The broader (and the more holistic) the valuation approach is, the lower is the probability to attain exact results in the details. Therefore, a tiered approach seems to be appropriate. The user must be aware of the uncertainties with respect to the applied instruments, and the decision maker must be informed about this insecurity.

Another next query might ask for the lessons learnt from the project experience. One summarizing answer is not possible here, because the contents would be much bigger than this book would allow. An attempt to concentrate some of the outcomes has been made in Chaps. 27–30 and several aspects of that outcomes have been discussed in this Chapter before. Here, we can add the demand for more applications of the described approach to understand, manage and protect human-environmental systems at the coast. Also the development of an integrated, generalized methodological platform seems to be an important target of future policy and science.

Q1: What can we learn from actual case studies of coastal ecosystem analysis in order to evaluate the actual condition of the ecosystems along the German Baltic Sea coastline?

Q2: Is it possible to integrate the multiple aspects of social, ethical and environmental sciences in order to characterize, indicate and measure ecosystem service potentials and flows?

Q3: Is such analyse a useful base for ecosystem management decisions and is it sufficiently significant, robust and applicable to serve as an instrument for sustainable policy?

Finally, we can now briefly answer the overarching questions Q1, Q2 and Q3 and initially formulate the statement that all participants have learnt a lot from the described case studies and that several pathways of environmental evaluation have been found, analyzed, compared and used. A summary of these items has been written right before. However, of course there are still several gaps of knowledge and technology. Some of these have also been listed before. They should be closed in order to consequently integrate the multiple aspects of social, ethical and environmental sciences to characterize, indicate and measure ecosystem service potentials and flows in the coastal environment. This can be a good basis for consequent applications of the approach in environmental management of human-environmental systems (Chap. 29).

Therefore, our suggestion is to continue with expanding ecosystem analysis to different media, to intensify the integration of human and environmental subsystems and to speed up developing integrated methods to find optimal solutions for forthcoming problems of the coastal environment.

Index

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