Developing the AQUACROSS Assessment Framework

Deliverable 3.2

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List of Acronyms

AF	Assessment Framework
BBNs	Bayesian Belief Networks
BD	Biodiversity
BEF	Biodiversity and Ecosystem Functioning
BES	Biodiversity and Ecosystem Services
CAS	Complex Adaptive Systems
CBA	Cost-benefit analysis
CEA	Cost-effectiveness analysis
CICES	Common International Classification of Ecosystem Services
CS	AQUACROSS Case Study
DA	Discriminant Analysis
DPSEEA	Driver-Pressures State-Exposure-Effect-Action
DoA	Description of Action
DPSEEA	Driver-Pressures State-Exposure-Effect-Action
DPSIR	Driver-Pressures State-Impact-Response
DPSWR	Driver-Pressure-State-Welfare-Response
EBM	Ecosystem-based management
EF	Ecosystem Functions
EPI	Economic Policy Instrument
ESS	Ecosystem Services
GDM	Generalised Dissimilarity Modelling

GDIM	Generalised Diversity-Interactions Modelling
GES	Good Environmental Status
GIS	Geographic Information System
GVA	Gross Value Added
IP	Information Platform
IPBES	Intergovernmental Platform of Biodiversity and Ecosystem Services
MA	Millenium Ecosystem Assessment
MAES	Mapping and Assessment of Ecosystem and their Services
MAUT	Multi-Attribute Utility Theory
MAVT	Multi-Attribute Value Theory
MPA	Marine Protected Area
MSE	Management Strategy Evaluation
MSFD	Marine Strategy Framework Directive
NWRM	Natural Water Retention Measures
SDMs	Species Distribution Models
SEBI	Streamlining European Biodiversity Indicators
SEEA	System of Environmental-Economic Accounting
SES	Social-ecological System
SUDS	Sustainable urban drainage systems
TEEB	The Economics of Ecosystems and Biodiversity
WFD	Water Framework Directive

About AQUACROSS

Knowledge, Assessment, and Management for AQUAtic Biodiversity and Ecosystem Services aCROSS EU policies (AQUACROSS) aims to support EU efforts to protect aquatic biodiversity and ensure the provision of aquatic ecosystem services. Funded by Europe's Horizon 2020 research programme, AQUACROSS seeks to advance knowledge and application of ecosystem-based management (EBM) for aquatic ecosystems to support the timely achievement of the EU 2020 Biodiversity Strategy targets.

Aquatic ecosystems are rich in biodiversity and home to a diverse array of species and habitats, providing numerous economic and societal benefits to Europe. Many of these valuable ecosystems are at risk of being irreversibly damaged by human activities and pressures, including pollution, contamination, invasive species, overfishing and climate change. These pressures threaten the sustainability of these ecosystems, their provision of ecosystem services and ultimately human well-being.

AQUACROSS responds to pressing societal and economic needs, tackling policy challenges from an integrated perspective and adding value to the use of available knowledge. Through advancing science and knowledge; connecting science, policy and business; and supporting the achievement of EU and international biodiversity targets, AQUACROSS aims to improve EBM of aquatic ecosystems across Europe.

The project consortium is made up of sixteen partners from across Europe and led by Ecologic Institute in Berlin, Germany.

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Foreword

Parmenides of Elea, the pre-Socratic Greek philosopher, explained in his poem *On Nature* two views of reality. In the so-called 'way of truth', he described how reality is one, change is impossible, and existence is uniform, timeless, necessary, and invariable. In the so-called 'way of opinion', Parmenides explained the world of appearances, in which one may be led to conceptions that are either false or deceitful. If avoiding self-indulgence, any reader may have noticed that when ignoring what to say, almost everyone would opine. And there is nothing wrong in making judgements (rather the opposite); the problem emerges when opinions are presented as scientific evidence.

Assessing is about evaluating, making (analytical) judgements or statements... it is not just about measuring, describing or informing. Hence, any Assessment Framework (hereafter AF) is a means to an end. Within AQUACROSS, the aim is to propose new ways of governing our relationship with nature (and not just with aquatic ecosystems, given the interrelationships with terrestrial ecosystems).

For that purpose, it is of paramount importance to understand processes and causes, rather than just describing and measuring states. The AF is a critical toolbox to that aim, grounded on conceptual considerations included in D3.1 (AQUACROSS Innovative Concept Note, Gómez et al., 2016).

The AQUACROSS AF deals with at least two main issues: *what* to assess (addressed in Part I); i.e where we are and where we could go through ecosystem-based management (EBM) approaches, against baseline, and *how* to assess it (see Part II). Unlike common wisdom, the emphasis for such an endeavour should not be on indicators. These are key and instrumental for the assessment (as will be reflected in Deliverables 4.1 (Pletterbauer et al., 2016) and 5.1 (Nogueira et al., 2016), developed in close coordination with this deliverable) but by no means are indicators the assessment itself.

For these purposes, as above, we build on the best available frameworks (see Section 1.2) but also harness state-of-the-art scientific knowledge. Building on other AFs does not mean at all assessing other frameworks. This sort of 'endogamic' exercise has actually been avoided as part of Deliverable 3.2 (D3.2). AQUACROSS is far from being about refining a tool whilst ignoring its actual use, but rather about shedding light on what is actually to be assessed and maximising the AF's practical usefulness to meet a number of objectives.

Needless to say that in order to build a new framework, to add value as part of this project, to innovate, the different communities of knowledge represented in the consortium have had to leave their "comfort zones" (both in terms of ecosystem-type and in relation to AFs they were previously familiar with). This effort, indeed, does require recognising the strengths of previous efforts but also their constraints, so as to integrate the former and overcome the latter.

Information systems, metrics, and descriptive efforts will be relevant but what is actually required for this AF is a more analytical view, on the basis of the best available scientific knowledge. D3.2 provides a comprehensive overview of these analytical approaches, following a logical sequence for the assessment itself.

But, what is the best available knowledge? A pervasive idea throughout the AF (and the project, as reflected for instance in the project's Science-Policy-Business Think Tank deliberations) is that we should find anytime the best knowledge available, no matter where it is. It is very likely that for ecological and social systems dynamics this will stem from science. Rather, for perceptions it may definitely come from stakeholders and for institutional barriers this is meant to come from experts (not necessarily scientists or stakeholders). Balancing this is a complex task and very much one that is at the core of what is to be done in AQUACROSS.

No assessment can be made in the absence of assessment criteria (see Section 2.3). On one side, evaluating baselines under well-defined criteria is key to identify threats to resilience, sustainability challenges, improvement opportunities, the suitability of the institutional set ups in place and the challenges to move away from unsustainable paths as well as to assess and eventually redefine policy targets regarding ecosystems and biodiversity (see Section 2.2). On the other side, assessing the outcomes of EBM approaches against baseline levels of biodiversity and ecosystem services (ESS) delivery is not something that can be tackled unless some explicit criteria are clearly defined.

Since AQUACROSS is about moving away from common practice, this AF has avoided, as much as possible, designing pieces even if we were aware they would not fit with each other, as well as driving the assessment towards either formal (i.e., compliance with EU Directives) or implied objectives (i.e., conservation of species and natural conditions). Conventional conservation efforts and compliance checking are full of the best intentions; this living document (D3.2), on the other hand, aims at making things happen. In accordance with the Description of Action (Task 3.2) the AF will now be tested in the different case studies and further developed as other work packages (WP1-WP8) evolve, leading to the final AF: D.3.3.

One may actually argue that subjective preferences and judgments are also part of assessing. As a matter of fact, perceptions are drivers of individual and collective actions, and are critical for both baseline and new policy scenarios (as in Section 2.1). Moreover, they are not just an input, if a critical one, but an opportunity. Projects like AQUACROSS, its scenarios and stakeholder engagement processes, could do a lot to change perceptions; that is to say, to enhance science-based perceptions, to make expectations from stakeholders compatible to each other, to enhance cooperation and policy coordination, to align incentives, etc.

What is assessed, as part of this analytical framework, is the new policy response (EBM approaches) against properly defined baselines. These baselines are a commonly agreed upon and shared representation of current and future problems, challenges and opportunities. These scenarios are necessarily co-built with stakeholders. An exercise of this kind should necessarily lead to an increased demand of accurate scientific answers to

relevant problems and may provide the basis for a common perception of the problem and its drivers, which in turn is a critical requirement for cooperation and collective action.

It is important to note that baseline and policy scenarios are the connections between analysis and policy, both of which crosscut throughout the AF. All models are available to assess and provide analytical linkages (as above, to explain rather than just to describe), then feed into the comprehensive assessment and scenarios of the system.

Needless to say that the discussion about policy objectives (see Section 2.2) entails making decisions about what should be part of the baseline and what should be part of the AQUACROSS policy scenario. Such a decision is not part of the definition of this AF, though, but rather of the actual assessment, to be developed in the different case studies.

1 Part I: What To Assess

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1.1 Introduction

1.1.1 The AQUACROSS Assessment Framework as an integrative and cooperative effort

AQUACROSS Assessment Framework (AF) combines scientific analyses to develop an integrative understanding on drivers, pressures, state of aquatic ecosystems, ecosystem services (ESS) and abiotic processes of ecosystems, and impacts on those ecosystems – ultimately on their biological diversity and ecosystem service delivery.

The emphasis, as in the Description of Action (DoA), is not on each one of those individual elements as such but rather on causal links between each one of them. It is not individual clogs of that logical chain the project focuses on, but on the complex linkages between each of them, following the AQUACROSS Concept (Deliverable 3.1: Gómez et al., 2016).

Literature shows that causal relationships are characterised by their strength (Yeung and Griffiths, 2015), consistency (Norton et al., 2014), specificity (Woodward, 2010), and temporality (Norton et al., op. cit.), but there seems to be consensus that only the latter is actually significant (Worm et al., 2006). The discussion about causality, though, alike any statement on association or correlation, will definitely emerge throughout the project when progressing from the design of this AF to its actual implementation in the different work packages and case studies. As below, this will be reflected in D3.3, aimed at providing an updated and upgraded version of the AF towards the end of the project.

Hence, the AF is the combination of those scientific analyses, stemming from different disciplines of knowledge and integrative efforts. This analytical framework should enable the practical application of EBM approaches in aquatic ecosystems through relevant models and guidance protocols, using adequate sets of data and indicators.

The AF will be tested, to different extents, in eight case studies and applied to a suite of innovative and applicable management solutions for aquatic ecosystems that serve to best enhance, through the conservation of biodiversity, the social–ecological resilience of the ecosystem and its capacity to deliver services to society. The project thus follows an 'idea to application' approach building on existing knowledge and generating innovative responses to policy coordination challenges by developing integrative tools and concepts with relevant stakeholders. Yet, the emphasis of this document, by definition, is not on the application of

the AF but rather on its design – ulterior work in the project will stress on the actual implementation of these analytical elements.

Integration (as well as inter- and trans-disciplinary research) is at the core of the AQUACROSS concept, thus making cooperative efforts not a choice but a logical need. The AQUACROSS AF is the outcome of these efforts, integrating across all aquatic ecosystems (freshwater, coastal, and marine) and mobilising expertise and knowlesge from biologists, ecologists, chemists, eco-toxicologists, hydrologists, oceanographers, environmental scientists, physicists, economists, IT-experts, and other social scientists. The resulting AF can be said to be integrative in different ways since it addresses:

▸ The harmonisation and streamlining of environmental policies under the overall framework of the EU 2020 Biodiversity Strategy and other international biodiversity targets.

▸ The coordination of policies in transitional and coastal waters, where different policy directives apply.

▸ The integration of relevant information for the assessment of aquatic ecosystems and their abiotic outputs across the freshwater-saltwater continuum.

▸ Social-ecological systems (SES) in a holistic way, as complex adaptive systems (CAS) that co-evolve, thereby avoiding traditional silos and biased approaches.

Central to AQUACROSS is the notion of EBM. EBM sets the foundations for the development of effective and widely applicable management concepts and practices for aquatic ecosystems. The EBM concept is concerned with ensuring that management decisions do not adversely affect ecosystem functions (EF) and productivity, so that the provisioning of aquatic ESS (and subsequent economic benefits) can be sustained in the long term. EBM is also relevant to maintain and restore the connections in SES, as well as a way to address uncertainty and variability in dynamic SES in an effort to embrace change, learn from experience and adapt policies throughout the management process.

This document presents a common framework for the assessment of aquatic ecosystems that is needed for the development of integrated management concepts. This framework is in line with existing assessment initiatives (see Section 1.2) and integrates ecological and socio-economic aspects in one analytical approach. Moreover, the AF considers relevant aspects for management of aquatic ecosystems specifically in relation to resilience and uncertainty. Yet, in addition to the theoretical underpinnings of the analysis, the AF also reflects a joint understanding of the key impacts on aquatic ecosystems between scientists, policy-makers and stakeholders and among ecosystem types (freshwater, coastal, marine), definitely benefiting from insights from a stakeholder workshop held in Berlin (March 2016) and the first AQUACROSS Forum held in Alcalá de Henares (Madrid, June 2016).

The AF helps facilitate the integration process of ulterior scientific work in the project's work packages by identifying available models and information as well as further data needs. Finally, the framework highlights key areas or 'nodes' where indicators are essential for

capturing the state and dynamics of biodiversity and ESS, as well as the adaptive capacity and resilience of SES. The framework, as a living document, facilitates synergies and identifies critical linkages between the different elements of the project: the analysis of drivers and pressures; the assessment of causalities between biodiversity, and EF and ESS, as well as their abiotic components; the impact of drivers on the status and trends of biodiversity, EF and ESS; the development of indicators to capture all relevant social-ecological-economic dimensions at the case-study level and beyond; and the design and implementation of EBM approaches, as innovative responses to enhance the status of aquatic ecosystems and achieve the relevant policy objectives at stake.

As in Section 1.2, recent years have seen a vast number of research initiatives promoting a range of concepts, methods, and models that aim to support the achievement of EU and international biodiversity targets. By explicitly considering the full range of ecological and human interactions and processes necessary to sustain ecosystem structure and functioning, EBM has become a most promising approach (Tallis et al., 2010), encompassing a whole range of decision support systems. Within that context, EBM has permeated, to a different extent, scientific and policy practice related to the management of aquatic ecosystems (Nobre & Ferreira, 2009).

While all those initiatives (including ongoing EU-funded research projects) provide a number of useful tools and products for decision-making, a major challenge remains in the establishment of an operational framework that links the assessment of biodiversity and ecosystem functions and services and their integration in public and private decision-making.

1.1.2 AQUACROSS Assessment Framework as a living document

The AQUACROSS AF (current Deliverable 3.2) has been developed as part of Task 3.2 of the project, building on Task 3.1 that led to Deliverable 3.1 (Gómez et al., 2016). Through further refining the concept and proposing potential methods and tools to be included into specific work-package research, where the AF will be tested in the different case studies, the AQUACROSS AF provides the foundations for applied research in the remainder of the project.

Through integrating the assessment of causal relationships between ecosystem functions and services and biodiversity levels in aquatic ecosystems in the conventional DPSIR (Driver-Pressure-State-Impact-Response) framework and overcoming some of its constraints, the AQUACROSS AF provides elements to assess, in sequential order both from a static and dynamic perspective:

▸ Drivers, pressures and multiple stressors (see Section 2.4), to better understand the sensitivity and dynamics of ESS to environmental change (and specifically biodiversity loss), as well as the environmental limits of ecosystems (i.e., threshold analysis). Behavioural models are reviewed in order to assess the implications of biodiversity loss and ESS delivery for human well-being.

- Understanding causalities, focusing on required elements for the quantification of the characteristics of biodiversity (from population to communities, habitat types, landscapes and seascapes) required for delivering ESS (see Section 2.5).

- Ascertaining impacts and responses to enhance the meaningfulness of some economic variables (value, price, cost estimates) in co-decision processes; to assess the added value of ecosystem-based approaches able to recognise the role of multifunctional land management and landscape and seascape patterns on the delivery of aquatic ESS and to develop options to enhance biodiversity levels and maintain ecosystem integrity beyond protected areas; as well as to promote the uptake of business opportunities associated with the sustainable management of flows (and stocks) of ESS (see Section 2.1).

Furthermore, the AF deals with several crosscutting issues:

- Resilience thinking, critical in the definition of scenarios (see Section 2.1.7) but also in the design and implementation of responses, to deal with uncertainty and to respond to unexpected changes (as these systems are characterised by non-linear dynamics, complex interactions across scale, self-organisation, etc.), through enhancing diversity and redundancy (not only ecological but also in the social system) and diversity of knowledge and response options and to provide opportunities for learning (e.g., in stakeholder processes) and changing policy directions based on new insights.

- Uncertainty linked to the assessment of information/data, and methods and tools required for creating scenarios of trends in drivers and pressures, causal links between biodiversity and ESS delivery, trade-offs between competing objectives, valuations, etc. (see Section 2.6.2).

- Dealing with varying spatial and temporal scales related to ecosystem function, services and human benefits, to progress towards adaptive responses (see Section 2.6.3).

- Data and metrics, reinforcing ongoing processes such as the reporting on SEBI (Streamlining European Biodiversity Indicators) and Aichi indicators, monitoring progress towards the EU2020 Biodiversity Strategy and other global targets, as well as ensuring coherence with other relevant policy processes (see Section 2.6.1).

Unlike other projects, AQUACROSS aims at continuously reviewing and refining this AF towards Deliverable 3.3 (Final Assessment Framework). For that purpose, the development of the AQUACROSS AF and, therefore, the investigation into the specific elements for assessment, are mindful of the practical challenges to be faced in terms of applicability (e.g., linking policy and science in the three aquatic realms); making the most out of existing knowledge to enhance current EBM practice; ensuring relevance (i.e., through making EBM truly operational in the three realms and in an interconnected way); etc.

As above, the DPSIR framework is used as a reference rather than a mould. D3.1 included a discussion of the strengths and weaknesses of that widely used sequence:

1 The DPSIR framework does not account for feedback processes.

2 It focuses on a single pressure, thus neglecting multiple stressors.

3 It does not allow for the discussion and assessment of trade-offs in terms of natural use, conservation, and enhancement.

4 The sequence is limited (or very limited) in linking human welfare and ecosystem functions and services.

5 It favours reactive and remedial responses rather than proactive and preemptive ones.

1.1.3 How the AQUACROSS Assessment Framework links to the different elements of the project

At the core of the project's research efforts, the AQUACROSS AF is linked to all the different work packages of the project, thus providing direction to the work to be developed throughout the project.

a) On one side, it is important to emphasise that stakeholder input is essential to support the deployment of the AF and its practical application in the different case studies. Innovation is about end-user driven research outcomes and therefore will be co-developed with stakeholders.

b) Through identifying policies affecting the achievement of EU and international biodiversity targets and assessing the operational demand for aquatic biodiversity, previous efforts of the project (as reflected in Deliverables 2.1 – *Synergies and differences between biodiversity, nature, water and marine environment EU policies: lessons learnt for coordinated implementation* and 2.2 – *Review and analysis of policy data and information requirements and lessons learnt in the context of aquatic ecosystems*; Rouillard et al., 2016 and O'Higgins et al., 2016 respectively) feed into the development of the AF.

c) The hands-on analysis of links between drivers and pressures builds on the AQUACROSS Architecture (see below) to develop in more detail the drivers and pressures dimensions of the AQUACROSS concept. The AF hence provides the basis for assessing the interaction between the full range of drivers and multiple interacting pressures and identifying sensitive indicators for the assessment of changes in ecosystem state for all aquatic realms (see Deliverable 4.1 – *Guidance on indicators, methods and tools for the assessment of drivers and pressures on aquatic ecosystems, including results from the meta-analysis*: Pletterbauer et al., 2016).

d) The analysis of causalities between biodiversity, ecosystem functions and services builds on the AF. This includes the development of methods and indicators for the assessment of causal links between diversity and aquatic ecosystem functions and services (see Deliverable 5.1 – *Guidance on methods and tools for the assessment of causal flow*

indicators between biodiversity, ecosystem functions and ecosystem services in the aquatic environment. Nogueira et al., 2016).

e) The AQUACROSS' Information Platform (IP) (forthcoming Deliverable 6.2 – *Development of the Information Platform*) is based on results, data compilation and assessments (including policy requirements and end-user's needs) that stem, among others, from the practical application of the AF.

f) Building on the AF, the research consortium will apply different modelling approaches to evaluate projected changes of drivers and pressures according to (participatory) scenarios across the different aquatic realms (forthcoming Deliverable 7.1 – *Guidance on methods and tools for the assessment of projected impacts of drivers of change on biodiversity, ecosystems functions and aquatic ecosystems service delivery; teaching modules for the individual modelling approaches*). This includes the use of probabilistic networks, species distribution modelling, social-ecological modelling, etc. and the development of guidance for implementation of the case studies to ensure consistent modelling across different realms.

1.2 Building on (and overcoming) previous assessment frameworks

Over the last few decades, a number of relevant AFs have been developed. Most of them formally aim at enhancing the understanding of the linkages operating in natural and social systems, and between both the demand and supply of ESS and abiotic components of ecosystems.

AQUACROSS' concept factored in their relevant elements to design the AQUACROSS AF. The concept (Gómez et al., 2016) largely acknowledges their strengths and limitations regarding its applicability in the three aquatic realms, going beyond to include dimensions that make them operational, such as multiple scales, a good science–policy–business interface and resilience thinking principles.

Table 1 summarises existing frameworks, focusing on what is assessed in each one, to draw a wide picture of relevant elements and to highlight those that might be useful to AQUACROSS:

▸ How are **drivers of ecosystem change linked** to **social and economic processes?**

▸ How are **adaptive ecological processes** resulting from **pressures linked** to **changes in the structure** and **functioning** of the ecosystem, and thus to the **delivery of ESS (ecosystem services)?**

▸ To what extent do those **social dynamics and responses** trigger those **changes**? Could we **prioritise** them regarding aquatic **ecosystem management**?

▸ What is the role of **biodiversity** in this **connection**?

Existing models (environmental, social and integrated models) could improve as they have built-in missing aspects, weaknesses in temporal, spatial or conceptual applicability or simplified structure. However they represent a meaningful starting point to match the analytical ambitions of the AQUACROSS project and to provide insightful and relevant explanations (rather than descriptions) on the casual relationships involved in the AQUACROSS Architecture.

Table 1: Comprehensive table of existing analytical or assessment frameworks

Framework	What is assessed	Usefulness to AQUACROSS AF	Constraints/ Difficulties to overcome/ Further contributions
DESSIN ESS Anzaldúa et al., 2016	Ecosystem Services (ESS)	Innovative solutions for water scarcity and water quality, impacts and benefits	Follows an adapted DPSIR Framework.
DPSIR, EEA, 1999, 2003	Linkages between ecosystem State and D–P, I–R	Causal links, Policy interface	Multiple pressures, interlinkages, non-linearities, synergies (Rekolainen et al., 2003; Svarstad et al., 2008) and societal/ecosystem responses have to be considered (Gari et al., 2015). Unclear boundaries/definitions (Cooper, 2013; EEA, 2015a; Gari et al., 2015). Limited focus on ESS (Collins et al 2011; Kelble et al., 2013). More references: Atkins et al., 2011; Kandziora et al., 2013, Impact as impact on human welfare, (Langmead et al., 2007; O'Higgins et al., 2014a), applied to freshwater (Koundouri et al., 2016), a review for coastal management (Lewison et al., 2016), DAPSI(W)R(M) for marine (Scharin et al., 2016). See also derived Driver–Pressure–State–Exposure–Effect–Action (DPSEEA: Reis et al., 2015), Framework for Ecosystem Service Provision (FESP: Rounsevell et al., 2010), Integrated Science for Society and the Environment framework (ISSE: LTER, 2007).
EPI–Water project, Zetland et al., 2011	EPIs (Economic Policy Instruments)	Environmental and economic dimensions, Coupled human and natural systems, Linkages through behaviour	No known constraints, difficulties to overcome or further contributions.
GLOBAQUA project Navarro-Ortega et al., 2015	Stressors	Water management options under scarcity	Limited to freshwater (river basins) and identified stressors.
IPBES, IPBES, 2014, Díaz et al., 2015	Social–ecological system (SES)	Linkages between biodiversity and ESS, Stakeholders, Knowledge – policy interface, Multiple scales	Testability. Shared-understanding (Potschin et al., 2016). Lacks a long-term scenario strategy (Kok et al 2016). See also Sub-Global Assessment Network (SGAN)

MAES Maes et al., 2013, 2016, Erhard et al 2016	ESS: mapping and valuing + data on ecosystem condition by EEA	Building baseline scenarios for supply side	Integration of linear river network data into spatial ecosystem maps (EEA, 2015a, 2016), Data for marine ecosystem and ESS considered for freshwater (Maes et al., 2013, EEA, 2015a). Demand side, monetary valuation (Brouwer et al 2013). Absence of stakeholder consultation to complete data and models at a lower spatial scale. See also ETC/BD: marine pilot study (Evans et al., 2014).
MARS project, Hering et al., 2015	Linkages between multiple stressors, ecological responses and functions	Water bodies, Long term, Informs EU water policies	No known constraints, difficulties to overcome or further contributions.
Millennium Ecosystem Assessment (MA) MA 2003, 2005	ESS	Conceptual Framework, Integrates different perspectives	Intermediate ESS are not considered and limited economic valuation (Brouwer et al 2013). Spatial and temporal data coverage. Models for long-term dynamics are required. Transition zones. Multiple scales and difficulties translating ecological to political boundaries and vice versa. Uncertainties developing scenarios. Another conceptual framework to analyse ESS delivery (Villamagna et al 2013).
MSFD, MARMONI Tool	Environmental Status	GES determination Indicators	Multiple components, double counting, comparability across regions (Borja et al., 2014; 2016). ETC/BD contribution to the state across marine regions (Aronsson et al., 2015). Requires intercalibration to demonstrate the coherence in application and implementation. See also MaPAF (Marine Protected Area Protection Assessment Framework, Rodríguez-Rodríguez et al., 2016).
ODEMM project, Robinson et al., 2014	EBM options, Impact chains, Weight of the links and threats	Holistic, trade-offs, Adaptable regional scales, Pressures and GES descriptors, Prioritizing EBM responses, Resilience	Limited to marine ecosystems. Knowledge gaps and data limitations. Limited applicability of criteria with available information. Non-linear effect can be undervalued. Tools: Assesing the State of Good Environmental Status; Linkage Framework; Pressure Assessment; Ecological Risk Assessment; Integrated Management Strategy Evaluation (iMSE).
OpenNESS project. Cascade model, Potschin et al., 2014	ESS	Operationalisation, Informs policy, Cross-scale analysis	Limited for decision-making (Potschin et al., 2014; Primmer et al., 2015).
OPERAs project, Kettunen et al.,	ESS	Demand side, Contributions to	See http://operas-project.eu/resources for relevant project deliverables.

2015		human well-being	
POLICYMIX project, Barton et al., 2014	Economic instruments for biodiversity conservation and ESS provision	Effectiveness, Efficiency/Cost-effectiveness, Legitimacy/fairness, Legal and institutional fit, Interaction between policies	Multi-scale approach is difficult.
RACER project, Christie & Sommerkorn, 2012	Ecosystems' features (drivers)	Region-wide resilience, Forecasts and scenarios, Management targets	Focused on climate change effects.
RAPTA O'Connell et al., 2015	Projects	Principles of resilience thinking, Understanding complex adaptive systems (CAS), Multiple social-ecological interactions and decisions	Limited to development projects.
Regulatory Impact Assessment (RIA) OECD, 2008	Policies	Cost-benefits analysis, Scenarios	Limitations linked to acceptance (OECD, 2008). Limited integration of ESS concept (Diehl et al., 2016). See also Integrated policy Impact assessment (IA).
SEEA UN, 2014: Central Framework	Stocks and flows		See also UN, 2012: SEEA for Water and Brouwer et al 2013 for a discussion.
SES framework (built on the foundations of the IAD framework; McGinnnis & Ostrom, 2014)	Social-ecological system	User's choices, Self-organizing, Understanding CAS	See also Ostrom, 2009 for applicability on fisheries management.
SOER project, EEA, 2015	Policy responses, trends, NC, state, impacts and scenarios	Societal adaptation, EBM approaches, Resilience thinking, Demand side (linkages to well-being)	Focused on emissions and pollutants. Applicability under national scale. Contributions of European Topic Centre on Inland, Coatal and Marine Water (ETC/ICM).

SRC, Schlüter et al., 2015	Social-ecological systems	Principles for building resilience: Simonsen et al., 2014, ESS included, CAS thinking, Policentricity	See also Rocha et al., 2015 for contributions with models on marine regime shifts.
Strategic Environmental Assessment (European Commission, SEA)	Plans/programmes	Provides scenarios	Limited flexibility and dynamism to be strategic enough for decision-making (Sheate et al., 2001, King & Smith, 2016).
TEEB, TEEB, 2010a	Economic and social drivers, ESS valuation	Understanding social choices, Addressing trade-offs	Valuation does not reflect variation in ecosystem quality. Intermediate ESS are not considered (Brouwer et al 2013). See also Russi et al., 2013 for Water and Wetlands. (See also: the *Ecosystem Properties, Potentials and Service* (EPPS: Bastian et al., 2013) framework differentiates between the potential and real supply of ecosystem goods and services and the biophysical prerequisites underpinning these).
UK NEA, UK National Ecosystem Assessment, 2014	ESS	Identified social values	Data limitations for a complete set of ESS. Heterogeneous valuation of ESS (Brouwer et al 2013). See also Kenter et al., 2013 valuing UK Marine Protected Areas (MPAs)
UNEP-WCMC, SANBI & UNEP-WMCM, 2016	Ecosystems (spatial assessment)	Informs policy, Prioritisation (NBSAPs), Guiding principles (usability, integration across realms)	Focused on threats and ecosystem condition on protected areas. Categorization of threat status of ecosystems (simplicity). It includes ecological processes and ESS, but not the demand side. Limited applicability on different spatial scales.
WFD Guidance, EC, 2003	Pressures and impacts. Linkages to ecological status	Integration, Stakeholders, Economic analysis, derived PoMs	Analysis follows DPSIR framework, with specificities for surface water bodies (river, lakes, coastal/transitional) and groundwater bodies. European Topic Centre on Biological Diversity (ETC/BD) contribution to integrated wetland management (Snethlage, 2015).
WISER project, Models, Methods database	Ecological status, recovery scenarios	Identifies degree/causes of degradation, Integrated perspective including uncertainty	Focused on restoration. Limited to freshwater.

1.3 What is to be assessed

1.3.1 Social-ecological aquatic systems as complex and adaptive ones

As stated in the AQUACROSS Innovative Concept (Gómez et al., 2016) the project's research strategy is based upon the consideration of SES as complex and adaptive, as well as mutually interdependent. As in Hagstrom and Levin (2016), over the last two decades complex adaptive systems have been refined from a somewhat abstract notion into a concrete concept with a series of tools and practical dimensions (i.e., coupling ecological and evolutionary dynamics, integrating multiple scales, using data to infer complex interactions, etc.), which can be used to address specific societal challenges.

Pondering the interdependencies between ecological and social systems does not seem to be merely an option anymore but a pre-condition both to better understand the social, political and environmental challenges we face and to compare alternative courses of action, whilst improving the collective capacity to respond to them.

Complex adaptive natural systems are characterised by emergent patterns, such as food-web structure and nutrient cycling. Two fundamental and intertwined sets of challenges are therefore to be understood: the first are fundamental modelling challenges presented by the interplay among phenomena at different scales (time, space, organisational complexity), as addressed in Section 2.6.3; the second involves the resolving the public-good and common-pool-resource conflicts that emerge.

The latter cannot be addressed at all without an equally complex understanding of not just the natural dimensions of these emergent patterns in ecosystems, but also of social and economic dynamics. The reason may seem obvious and still remain unaddressed in conventional approaches: these challenges arise not only in the description but also in the management of any complex adaptive system. The AQUACROSS AF explores them in aquatic ecosystems.

AQUACROSS' integrated approach to sustainability, as reflected in this AF, thus builds upon the understanding of both systems and their interlinkages to develop innovative management approaches and tools focused on the restoration and protection of critical aquatic ecosystem components, as a means to sustain biological diversity and the delivery of ESS in the long term.

1.3.2 The integration of knowledge as a means to truly integrated responses

According to the innovative concept (*op. cit.*), at the analytical level AQUACROSS aims at mobilising and integrating knowledge so as to understand 1) how social and ecological

systems are linked at multiple levels and across different scales, 2) how these linkages give rise to the dynamics we see at the system level, and 3) what the role of mutual adaptation/co-evolution is.

Yet, all this should be functional to a more specific objective on practical grounds, consisting in providing the means and the ends to deliver a better political response to current sustainability challenges in all policy domains linked to water and biodiversity. Full scenarios instead of bounded models are therefore critical given the emphasis on showing the advantages of holistic approaches over partial ones.

Dietz et al. (2003), in their seminal paper on the struggle to govern the commons, highlighted that no challenge facing ecosystem science is actually more important than managing the interactions of humans with their environment. Not in vain, some of the central features of complex adaptive systems (CAS), such as conflicts between individual and collective goals or alternative stable states, are common drawbacks to the management of natural resources and SES.

Hence, the ambition of improving knowledge to provide better management responses pervades the full strategy of the project from the definition of policy challenges or the setting of objectives, to the identification of opportunities, the screening of innovative responses and the design, implementation and assessment of alternative courses of action. All these elements must be framed into comprehensive and holistic frameworks able to capture all relevant interactions at stake, thus making the difference with limited partial approaches more visible.

Realising this ambition entails new requirements in terms of analytical complexity. For instance, the definition of any policy problem requires factoring in multiple interactions and not just the set of variables more directly linked to a well-defined target, such as reducing overfishing of particular species, reducing water demand up to a certain level or guaranteeing a certain water quality parameter.

Rather, AQUACROSS aims at showing the shortcomings of prevailing practice, both in collective and private decision-making frameworks, consisting in managing the provision of particular ESS (such as water supply, fish biomass, timber), through controlling others (water storage, provision of food and raw materials) at limited temporal and spatial scales (mostly short term and local) within well-defined system boundaries (of a water body, fishery or a plantation) and assuming the stability of the system throughout the medium and the longer term (see Gómez et al., 2016, Section 2 for more details). In other words, in order to fulfil its research ambitions, policy problems cannot be set within the boundaries of partial optimisation models but rather should rely on the best possible understanding of the whole SES.

Overall, instead of specific optimisation models, comprehensive scenarios that are able to inform decision-making are required. This is consistent with the project's hypothesis that decision support models that do not consider the complexity and adaptability of natural and

social systems may well lead to socially and environmentally irrelevant (or, at worse, undesirable) outcomes.

Models that exclude non-linearities or even the possibility for evolution and adaptation still typically guide aquatic ecosystem management. Policies and plans are often thwarted, thus leading to critical transitions and irreversible losses. Very often this is explained as a result of the unpredictability of human behaviour or the intricacy of the underlying ecosystem dynamics, when in fact it is likely to be the outcome of an inadequate assessment of both issues and their complex interactions.

1.3.3 Co-building scenarios between stakeholders and researchers: business as usual versus new policy responses

The AQUACROSS baseline scenarios are built to provide a comprehensive representation of the overall SES focusing on the relevant interactions and identifying environmental and policy challenges. This effort combines scientific knowledge and data with stakeholders' perceptions. Therefore, building a baseline scenario is not just a scientific endeavour but also the result of matching this knowledge with expert judgements and stakeholders' beliefs and perceptions (Caudron et al., 2012).

Ideally, a baseline scenario is a commonly agreed upon and shared representation of current and prospective problems, challenges and opportunities that society and the environment face (Verburg et al., 2015; Pichs-Madruga et al., 2016). For this reason, the right approach consists in co-building that baseline through a meaningful science-policy dialogue in which, for instance, first impressions by stakeholders on the factors driving ecosystems' degradation are challenged with empirical evidence and scientific explanations.

An exercise of this kind may result in the demand of precise and well-focused scientific answers to relevant problems (such as whether reductions in fish biomass landings are due to previous overfishing or to the degradation of the supporting ecosystem). Such an exercise may also provide the basis to build a shared perception of the problem and its driving factors, which is a critical requirement for cooperation and collective action.

The design of scenarios plays a critical role in the entire project at least for two key reasons (see Prewitt et al., 2012, for an insightful discussion along these lines):

▸ It allows the enhancement of the policy relevance of scientific knowledge. Well-designed scenarios are but communication platforms that bring science into the policy-making process, thus making stakeholders aware of multiple relevant interactions in SES and helping them assess current practice, screen new opportunities, and improve the design and implementation of policy responses. Policy-relevant, scientific knowledge makes the value of science for policy visible and allows aligning research, innovation, and policy priorities.

- In turn, it fosters the enhancement of the scientific foundations of policy. Frontier (or simply new) knowledge is functional to the identification of novel courses of action. It also favours a better identification of the opportunity costs and the benefits of traditional and innovative approaches. In such a way, it is therefore possible to anchor many sensitive policy debates on empirical evidence, instead of just on perceptions or prejudices. This leads to shared views of sustainability challenges amongst stakeholders so as to promote cooperative responses rather than competitive ones. Policymaking based on scientific knowledge supports a common ground and helps build consensus, hence focusing policy discussions on trade-offs and making choices where stakeholders' preferences and vested interests are really relevant.

Box 1: Scenarios and the policy-making process

Ferrier et al. (2016) identifies four types of scenarios depending on their role over the policy-making cycle:

1. **Baseline scenarios** that represent observed past and plausible futures, often based on storylines and on the best information available (as the ones considered in Section 2.1).

2. **Normative or "target-seeking scenarios"** representing objectives, deficits and sometimes alternative pathways for reaching this target (as the ones considered in Section 2.2 on policy objectives).

3. **Policy-screening scenarios"** (also known as 'ex-ante scenarios'), to represent, assess and compare alternative policy instruments or measures ex-ante.

4. **Retrospective policy evaluation** (also known as 'ex-post evaluation'), that represents the observed trajectory of a policy implemented in the past and assess by comparison against baseline scenarios.

On a similar note, the analysis of scenarios involves two main intertwined objectives:

- The first general objective is a *positive* (as opposed to normative) one. It consists in representing the best available knowledge to understand the complex SES to be managed. At this stage the main purpose is making the AQUACROSS concept and Architecture (see below) operational so as to provide stakeholders with the very best science to understand management challenges and opportunities at hand and to help them build a shared perception of the problem, and of the alternatives to deal with it. Knowledge about the different parts of ecosystems and society is piecemeal (i.e., not all parts of knowledge are available) and clearly imperfect (i.e., science is imprecise and most relationships are, to some degree and at best, uncertain). This is against the ambition of building an extensive representation of the whole SES and all its interactions. And this is also why building full-fledged science-based scenarios is an elusive task.

  Rather than a precise cookbook, the AQUACROSS Architecture, or the structure upon which the scenarios are built, consists in a heuristic approach based on the best available science but also on narratives and explanations to navigate through social and ecological interactions. Scenarios are informal but meaningful constructs that use all sources of knowledge, from hard science to narratives based on stakeholders' perceptions. Albeit

imprecise, their objective is to provide policy-makers with a comprehensive view of the SES.

▸ The second general objective is actually a *normative* one. It consists in assessing the whole system and representing the policy challenge at stake. Therefore, the baseline must serve to identify sustainability concerns. They include the identification and explanation of the underlying causes of the detrimental processes affecting the ecological system, and the socio-economic system (such as water depletion, biodiversity losses, population decline, increasing drought risk, etc.). Explaining the factors that drive these processes (such as wrong incentives, market conditions, inappropriate policy responses, etc.) and the need to take steps to handle these challenges (in order to curb degradation processes, protect human welfare, create job opportunities, avoid an economic downturn, etc.).

These problems must be presented as a governance challenge, that is to say a challenge requiring some sort of collective action to restore the consistency between private (individual) decisions (of those who benefit from existing ESS, such as fish or water provision that might not be used sustainably), and the public trust (that would presumably be preserved should current degradation processes come to a halt).

Both objectives (building a baseline scenario, representing the system, and ascertaining the management challenge) are closely connected to each other. For instance, the levels of detail, the activities or links that deserve more attention, the scales of the analysis, etc. are largely determined by the environmental challenge at hand. For example, baselines to inform freshwater management issues at river basin scales are indeed different from those required to support marine or coastal ecosystem management (see e.g., Ellis et al., 2011, including a case study in the Baltic Sea).

1.3.4 Ecosystem-based management approaches taking centre stage

The AQUACROSS EBM approach will be primarily driven by policy objectives (see Section 2.2). Overcoming current practice and progressing towards the implementation of holistic and integrated responses based on the ecosystem is significantly more demanding both for science and policy than 'going with the flow'.

On one hand it requires considering the complex structure and links of the full SES that have traditionally been ignored in conventional practice (see above). On the other it requires institutions and policy-making processes that are able to enhance cooperation and provide integrated responses to both the social and ecological challenges.

The concomitance of public-good and common-pool-resources attributes is inherent to CAS. Yet, so is cooperative behaviour, from microbial ecosystems to human societies. Understanding the enabling conditions that promote or the factors that hinder cooperative behaviour is critical to the design and implementation of EBM approaches: progress in

evolutionary theory (Wasser, 2013) and game theory (e.g., Punt et al., 2014) has been considered as specially promising.

Setting up social-ecological policy challenges and appraising the many different alternatives to cope with them is more than just a purely scientific or political endeavour. This ambitious task requires both a conducive political process and the mobilisation of the best available scientific knowledge to support stakeholders and decision-makers throughout the whole policy cycle.

Scenarios are double-edged decision-support systems: on one side, they must rely on validated data and sound scientific insights as a critical condition for their credibility but, on the other, they must have the ambition to become a collective representation both of social and ecological problems and opportunities and alternatives to deal with them. In fact, even a sound scenario based on scientific methods and proven facts would only be relevant for policy action if co-developed or assumed by social agents.

Due to the holistic nature and complexities involved in aquatic SES, it is clear there is neither a one-size-fits-all EBM approach nor just one EBM implementation path. Additionally, it is critical to understand that more science may not necessarily close the existing knowledge gaps. Rather, each individual situation (i.e., case study) may need to be considered in its institutional and political setting, and requires site-specific trade-offs.

Under an EBM approach science is not only intended to inform and make technically sound decisions but rather as a means to build a credible knowledge base through the dialogue and interaction between scientists and stakeholders. This involves the integration of multiple kinds of knowledge ranging from hard science to storylines.

It is often argued that EBM approaches are characterised by their contribution to ecological integrity, biodiversity, resilience and ESS delivery; their use of scientific knowledge; their use of appropriate spatial scales; their acknowledgement of social-ecological interactions, stakeholder engagement and transparency; transdisciplinarity and integrated management; and adaptiveness (see, for instance, Deliverable 2.1: Rouillard et al., 2016). However, unlike common wisdom, EBM does not exclusively show those features and several approaches that are not based on the ecosystem may well do.

EBM, though, of course shows some distinctive features:

▸ EBM factors in the dynamic to balance ecological and social concerns. EBM gives prominence to governance and relationship among and between species, as well as their abiotic environment.

▸ Unlike conventional approaches that focus on single benefits, EBM approaches are characterised by multiple benefits or environmental services, thus meeting at once targets across different policy domains. In other words, EBM aims at maximising the joint value of all ESS and abiotic outputs, rather than focusing on the delivery of single ESS.

- EBM approaches, like other management approaches, are based on scientific knowledge, but what sets EBM apart is the kind of scientific knowledge that is harnessed, as well as the way in which it is integrated into the decision-making process.

- Managing ecosystems is much more elaborate than managing single water bodies, single natural assets or even watersheds, just to mention a few examples. EBM decisions should therefore take place at the appropriate level, taking into account ecosystem boundaries, complex connections, and adaptive processes.

- The analysis of EBM may benefit from the exploration of the concept of meta-ecosystem (Loreau et al., 2003). This notion provides a powerful theoretical tool to ascertain the emergent properties that arise from spatial coupling of local ecosystems, such as global source-sink constraints, diversity-productivity patterns, stabilisation of ecosystem processes, and indirect interactions at landscape or regional scales.

- Ecosystem connections within and across aquatic realms should be considered, as management interventions in ecosystems often have unknown or unpredictable effects on other ecosystems.

- Rather than treating society and the environment as separate entities, EBM acknowledges social-ecological interactions and seeks inclusive policy-making processes that favour transparency and provide a better framework to confront people, businesses, and governments with the consequences of their own decisions.

1.3.5 The identification and structuring of policy objectives and the clearcut distinction between objectives and assessment criteria

The definition and structuring of objectives, essential for the assessment, builds upon the baseline analysis (see Section 2.1), where the main challenge and the policy context is to be set along with policy priorities for the local level. The definition of objectives and their operationalisation for assessing progress at the local level would benefit from the analysis of social drivers of ecosystems change, the resulting pressures and the assessment of the current, and baseline status of the relevant ecosystems (see Section 2.4), as well as from the analysis of how all this connects with biodiversity, and ecosystems services (see Section 2.5). The precise definition of objectives should provide a standpoint for screening, assessing, designing, and implementing the management alternatives to reach these objectives.

The identification of objectives combines two important levels that are complementary and closely connected to each other but clearly different in nature.

- At a global and EU level, objectives need to be defined in terms of contributions to meeting the targets of the EU 2020 Biodiversity Strategy and other international targets within aquatic ecosystems, while contributing to the objectives set in the EU directives and strategies related to habitats, biodiversity, and aquatic ecosystems.

- At a local level, objectives need to be defined to respond to a well-defined environmental challenge (such as dealing with invasive species, reducing nutrient pollution, improving hydrological flows and water retention, etc.).

These levels do not refer to different objectives but rather to how abstract EU-level goals are defined and specified at local and ecosystem scales so that, besides compliance with EU regulations, policy priorities, available information and local circumstances are taken into account.

The future of water and biodiversity depends on the concerted action of many agents from local to global levels. Global, regional or national actions are considered as part of the baseline scenario, while the policy scenario is centred only on those actions that could be adopted by local authorities and stakeholders within their powers and opportunities.

The AQUACROSS concept (Gómez et al., 2016) stresses upon the fact that both levels of objectives refer to desired or target conditions of the ecological system (rather than the SES as a whole). At a local level, however, objectives must be designed in order to restore the sustainability of the whole SES. This overarching goal entails a necessary precondition: reaching the sustainable status of the ecological system. It is now common practice that the goals of EU environmental policy (see Section 2.2) and the goals of the relevant strategies or Directives are stated in terms of conservation, protection, enhancement of biodiversity, habitats, water bodies, etc. In other words, whilst the assessment refers to both the ecological and the social system along with the complex links among them, primary objectives only seem to address the ecological system.

One may wonder why important societal objectives such as enhancing adaptability, improving the institutional capacity to design and implement comprehensive and ambitious EBM approaches, gaining political acceptance, improving fairness and other social goals are not the chief objectives of policy. The basic reason for this is that these ambitions are not objectives themselves, but rather the means that would make it possible to meet primary objectives of policy action in the domains of water and biodiversity.

This caveat is particularly relevant to clear out the difference between objectives and assessment criteria. The former (Section 2.2) refers to the primary ends of environmental policy whereas the latter (Section 2.3) refer to the criteria to judge the system and the alternative means that may be used to reach those goals. Within the AQUACROSS approach any policy objective is defined in terms of a desired or target condition of the involved aquatic ecosystem, including its biodiversity. Thus, the analysis of any other ambition related to the social system (such as mobilising enough financial resources, gaining political acceptance, improving social fairness, etc.) is considered within criteria to assess the alternative ways to reach the primary environmental targets, then to assess the institutional capabilities to meet what is actually required for sustainability.

1.3.6 A three-tier assessment on the basis of different criteria

Judgement criteria are connatural to policy action. The AQUACROSS concept and Architecture allows mobilising and representing scientific knowledge, data and models, in a comprehensive way such that this can be taken by stakeholders so as to improve policy-making and management decisions to address environmental challenges linked to water and biodiversity. Harnessing this knowledge to make policy decisions requires assessing current and prospective scenarios, as above, in order to evaluate the pros and cons of taking remedial actions, solving trade-offs, comparing alternative courses of action, improving policy design and implementation, etc.

The criteria under which alternative states and courses of actions are assessed are not only essential to make policy decisions but also to disclose alternatives in a structured, accountable, and transparent way. When those criteria are applied through the use of the best available knowledge, they reduce discretion in policy decisions in turn increasing trust amongst stakeholders; contributing to develop shared visions of environmental and economic challenges; and enhancing opportunities to improve cooperation.

Informed judgements, that is to say applying criteria for comparison, are inherent to decision-making. There are two complementary but closely connected levels of assessment that require differentiated criteria. On the one hand, criteria are essential to evaluate, or assess, situations or scenarios; on the other, they are key to assess different policy alternatives.

In the first case all criteria are functional to judge the sustainability of the whole SES. In the second what is important is a set of criteria to build up the comparison between the outcomes associated to alternative policy decisions (including inaction – an alternative also entailing costs and benefits). Accordingly, it is useful to make a clear distinction between two types of criteria: those designed for assessing the whole system (or *system* criteria) and those designed for assessing the outcomes of alternative courses of action (*output* criteria).

Emphasis will be placed on setting what criteria are relevant at any stage of the analysis, making a clear distinction between those that are most informative for judging the following three aspects:

▸ **Baseline scenarios.** Assessment of baselines is essential to identify sustainability problems, representing ongoing processes, supporting the definition of current sustainability challenges, and helping define policy targets at the scale of any study site. Assessment of baselines is also of paramount importance to highlight opportunities and barriers to overcome sustainability challenges, hence to support the definition of management strategies.

▸ **Policy scenarios.** Counterfactual scenarios result from the implementation of EBM approaches and must be judged, in general, on the basis of their contribution to sustainability and, in particular, for their intended and realised contribution to reach

policy targets (effectiveness) and for their contribution to human well-being (efficiency, equity).

▸ **Decision-making processes.** These processes refer to the potential of current institutional policy set-ups to properly address sustainability challenges within prevailing governance structures. It involves criteria for judging the capacity to overcome institutional inertia, technology lock-ins as well as conventional analytical approaches in order to advance towards better policy coordination, and innovative technological approaches based upon integrative management strategies (i.e., EBM). Criteria under this category will support reform efforts as an integral part of EBM.

1.3.7 The demand for aquatic ecosystem services delivery and abiotic outputs

For an improved understanding of aquatic SES and its interconnections, the AQUACROSS architecture (see below) considers two interrelated sets of linkages between the ecological and the socio-economic parts of the system. The supply-side perspective (see Section 2.5) describes and analyses the capacity of the ecological system to fulfil the social demands of ESS, thus contributing to human welfare. The demand-side perspective (see Section 2.4), in turn, describes and analyses how the effective demand of all kinds of ESS and abiotic outputs by the socio-economic system affects the ecological system, its structure, and functioning.

Accordingly, the demand-side perspective conceptualises how human activities result in demands of ESS and abiotic outputs that may trigger detrimental changes to ecosystems through the pressures they exert over their components and structure. Besides its relevance to understanding impacts of human decisions and actions over nature, this assessment level is also essential to understand how human action impacts ecosystems and biodiversity and the capacity of aquatic ecosystems to continue providing the services society depends on.

The link between society and ecosystems is analysed through the identification of all relevant social, policy and economic processes which may result in a pressure (or a combination of them) over the ecosystem or, in other words, of the drivers of human pressures over ecosystems.

The emphasis on human drivers within the overall AF is explained because these drivers and all their determining factors in the social and economic system can be the focus of management decisions. In other words, they can be changed. However, natural adaptive processes are indeed also explored as they are essential to build robust and realistic scenarios for evaluation of management strategies but their direct control is out of reach for humans.

Social or human drivers of pressures over ecosystems are the effective demand for all kinds of goods and services provided by nature to the social system, including ESS and abiotic outputs from the ecosystem. The basic goods and services provided by nature (such as raw water, fish, building materials, navigation, pollution disposal, etc.) are essential means to

produce a plethora of final abiotic outputs and services (such as drinking water, food, shelter, recreational services, clean air, health, etc. In turn, these products and services provided by nature are the actual reason human socity is concerned with the conservation of the ecosystems on which the future availability of these services, and thus human wellbeing, depends.

Within the AQUACROSS, drivers are the outcome of complex institutional, social and economic processes. Consequently, managing responses should go beyond the direct regulation of single activities (such as fishing) or related pressures (such as seafloor abrasion) to encompass broader management alternatives such as managing food chains, aquaculture, marine protected areas, incentives, pricing regulations, research, technological development and innovation, etc. For such ambitious management responses, having the best understanding of what determines the drivers is at least as important as describing the drivers themselves (see for instance: Martín-Ortega et al., 2015; OECD, 2016).

Pressures, in turn, are mechanisms through which a driver has an effect on the environment. Pressures can be physical (e.g., extraction of water, emission of noise), chemical (e.g., emission of chemicals or waste) or biological (e.g., extraction of aquatic species, introduction of microbes and non-indigenous species, etc.). These kinds of pressures are the direct result of primary activities to co-produce the nature-based services demanded by the social system. The pressures are different to those changes that are inherent to the processes taking place spontaneously in the ecological system (natural disturbances). However, to understand the complex mechanisms that lead to specific states of the environment or the ecosystem, they can be considered. Moreover, in the presence of those natural disturbances, pressures can exacerbate changes and push the ecosystem towards a regime shift, i.e. push the subsystem into a new stable state that is different to the former one, or to the acceleration of other change processes (Folke et al., 2004).

Within AQUACROSS, significant pressures are those that result in a change in ecosystem state leading to a change in the functioning of the ecosystem, and thus can impact both biodiversity and human welfare. Most studies to date attempt to deal with single pressures; yet, attempts have been made to consider multiple pressures and their cumulative or interacting effects on ecosystem state through additive, synergistic or antagonistic responses (see, for instance, outputs of EU FP7 MARS project).[1]

Understanding how ecosystem states change in response to human activities and their resultant pressures requires a good conceptual basis that links the causes and consequences of that change (Borja et al., 2016). This conceptual basis is most often described in aquatic realms in terms of a categorisation of information to capture multiple causes and the nature of change in ecosystem state, and the impacts of change on human welfare (Cooper, 2013).

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[1] MARS project: Managing Aquatic ecosystems and water Resources under multiple Stress

1.3.8 The supply of aquatic ecosystem services and abiotic outputs and how they are determined by biodiversity levels

Change in the state of aquatic ecosystems (i.e., changes in their structure and functions) can lead to changes in the supply of ESS and abiotic outputs, and thus in the services and the benefits to society that can be obtained, while at the same time compromising the preservation of ecosystems themselves.

The supply side influences the demand-side perspective in terms of how the ecosystem benefits society, but it is also influenced by the demand side through drivers, pressures and resulting changes in the state of the ecosystem.

The effects of pressures on the ecosystem have been explored both through field-based observations and experimental manipulations. These studies tend to inform about the effects at the species or, less frequently, the process level. However, it is relevant to understand how or if these changes would lead to any change in the capacity of the ecosystem to supply services. The metrics used to describe how pressures change ecosystem state may not be the appropriate metrics themselves to describe and explain how the ecosystem contributes to the supply of all services.

In order to consider how these changes might lead to an effect on the supply of ESS two elements are needed. Firstly, which services are underpinned by the functions and processes of benthic communities (flora and fauna)? Secondly, in what way these communities supply services and to what extent measurements of abundance and/or biomass capture this?

Pressures can have multiple effects and act on structures and processes and functions. Different services may be affected in different ways by the same pressure. Pressures can have direct and indirect effects on service provision. The way pressures affect the system is explored further in Deliverable 5.1 (Nogueira et al., 2016).

1.3.9 From datasets to data flows; moving from measurement to analysis

Understanding and predicting the behaviour of complex adaptive SES involves integrating information from many different disciplines. While the AQUACROSS Architecture (see below) (and other similar conceptual frameworks such as the DPSIR) provide a conceptual basis for broadly understanding the causal relationships between different components of the system, quantifying these interrelationships between different components requires specific disciplinary inputs, and may involve consideration of tipping points and non-linearities.

For any individual system there may be many environmental or social processes that remain unknown and non-modelled and which lead to a variety of uncertainties in the outputs of a

particular analysis. Within the AQUACROSS project the AF provides a basis for analysis of the various unique case-study systems.

The particular types of data and information for analysis in these specific case studies vary, as do the specific levels of disciplinary capacity and expertise. Designing a framework that is sufficiently flexible to accommodate the intricacy of adaptive SES, while also accounting for the variety and uncertainty in the types and quality of data for analysis therefore represents a major challenge.

1.4　How to read this document: What you will find and what you won't

Part II of this living document focuses on the conceptual and practical dimensions of how to assess EBM approaches (as part of new policy responses), against baseline scenarios, aiming at achieving the targets of the EU 2020 Biodiversity Strategy and other international biodiversity targets.

The document thus follows a sequential approach from the design of baseline and policy scenarios (Section 2.1) to the assessment of crosscutting issues (Section 2.6) going through the identification and structuring of policy objectives (Section 2.2), the definition of assessment criteria (Section 2.3), the assessment of drivers and pressures (Section 2.4) and the causal relationships between biodiversity, ecosystem functions (EF), and ESS (Section 2.5).

1.4.1　Open questions to be assessed throughout the project

The practical application of the AF should shed light on a number of questions (this list is not comprehensive):

▸ What are the most relevant drivers affecting aquatic ecosystems? How can the demand for ESS be compared against the ability of aquatic ecosystems to deliver services in a sustainable way?

▸ Are there alternative definitions of drivers and pressures depending on whether the anchorage is on science or policy?

▸ How to move from descriptive to more analytical approaches, given the challenge of having an abundance of information to fill into the different layers but relatively scarce amount of information to understand links between drivers and pressures.

▸ To what extent can knowledge on biodiversity loss drivers and indicators be adapted, downscaled and made useful for specific applied assessments (i.e., case studies)?

▸ How could one adddress the connection between the analysis of drivers and pressures and the ecological assessment of links between ecosystem functions and services and biodiversity through the assessment of changes in the state of aquatic ecosystem?

▸ How to go beyond the emphasis on indicators (and the constraints of modelling efforts) to analyse causal links between biodiversity and ESS delivery?

▸ Could a convincing storyline about those links (drivers, pressures, biodiversity, ESS, etc.) be built with no evidence about said links? Should this approach be caveated?

▸ How could the application of the AF shed light on the critical differences between causality and correlation, prediction and forecasting, statistical analysis and scientific knowledge? How can we move from predictive models towards decision-support tools measuring uncertainty?

- Could the outcomes of the analysis of the above-mentioned causal links be used to actually assess the effectiveness of policy options regarding biodiversity?

- How are existing models able to incorporate (if at all) EBM?

1.4.2 Several knowledge gaps

It is often overlooked that the adaptive potential of SES is essentially a constructive process, sometimes yielding persistent structures through processes of self-organisation, with the outcomes potentially uncertain and influenced by path dependence, including initial conditions. Aquatic ecosystems hence may exhibit alternative stable states, which suggests that the emphasis of the assessment should fall upon the robustness and resilience of particular configurations, and the potential for sudden shifts (including irreversible events).

The interplay between processes on different spatial, temporal and organisational scales, central to the remit of AQUACROSS, is also key to deal with CAS. This includes the emergence of regular patterns, the potential for regime shifts, and conflicts that arise between drivers of ecosystem change and the actual outcomes of pressures stemming from those drivers.

It is important to highlight that the emphasis should be on management-driven assessment outputs. This entails the need to address upscaling issues, as well moving from the local to the global dimensions of biodiversity targets, and from the short term to the long term.

Significant progress has been driven by interdisciplinary research in the past (and also within the AQUACROSS consortium). This creates good conditions to better understand critical phenomena and thresholds and the emergence of macroscale structure from microscale interactions. Yet, this remains a major challenge. So is the analysis of non-linear dynamics and stochasticity or the understanding of tipping points or the operationalisation of resilience of SES to external perturbations.

Clear knowledge gaps apply to the understanding of the inherent trade-offs between different social-ecological strategies. This AF will try to address these issues on the basis of illustrations provided by the different case studies.

2 Part II. How To Assess

2.1 Framing the decision context: baseline scenarios

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2.1.1 Introduction: why scenarios are critical

This first chapter of Part II of the AF mainly focuses on the two objectives mentioned in Section 1.3.3 above: representing the system on one side and setting the decision context on the other.

This section briefly discusses the use of qualitative and quantitative scenarios to assess trends in aquatic ecosystems both in terms of drivers and multiple pressures, and also impacts (i.e., changes in the ecological status) and responses.

When building such scenarios, a number of major shortcomings arise:

▸ For instance, regarding marine ecosystems, the absence of scenarios that assess the combined impacts of a changing climate, land-based drivers and pressures, offshore drivers and pressures, and overall socio-economic factors affecting the marine environment.

▸ Additionally, in marine ecosystems, the embrionary status of marine spatial planning issues, hence leading to lack of good datasets for spatial analysis.

▸ Overall, the lack of quantitative scenarios dealing with the link between drivers and pressures and the ecological status of aquatic ecosystems (including causal links between biodiversity levels and delivery of ESS and abiotic outputs).

▸ The fragmented approach to the freshwater-coastal-marine continuum, hindering the possibility of a more complex (i.e., realistic) approach.

▸ The weak consideration of governance and economic incentives in prospective efforts.

To address the needs of a broader audience (such as that targeted by AQUACROSS), which is too often unaware of aquatic ecosystems and their policy challenges, the scenarios may need to be more comprehensive than previous scenarios, as well as covering basic issues related to the good ecological or environmental status of the different aquatic realms.

The rationale for a long-term perspective lies in the need to take account of the slow unfolding of aquatic ecosystems and socio-economic processes and the necessary time for responses (e.g, Marine Protected Areas (MPAs) and other EBM approaches, biodiversity conservation target setting, marine spatial planning, etc.) to yield efficient, equitable, and sustainable outcomes.

This does not preclude at all the challenges to be faced when developing such a long-term perspective. As above, one may feel tempted to use forecasting techniques to estimate different futures for European aquatic ecosystems. Yet, although forecasts may be reliable in the short run, they would become more uncertain as the time span of the assessment expands.

Forecasts necessarily contain a fundamental uncertainty, based on our bounded understanding of social and ecological processes and the fact that aquatic ecosystems are inherently complex dynamic systems. In addition, there is also the fact that the future of these aquatic ecosystems largely depends on human decisions. Building scenarios is essentially a response to these uncertainties. A scenario is much more than delivering projections, forecasts or predictions (i.e., estimates) (see Figure 1). Scenarios should include a storyline (a hypothetical sequence of events) with a logical narrative about the way all the events in relevant SES may unfold to focus attention on causal processes and decision points.

Figure 1: The trade-off between uncertainty and complexity in forecasting
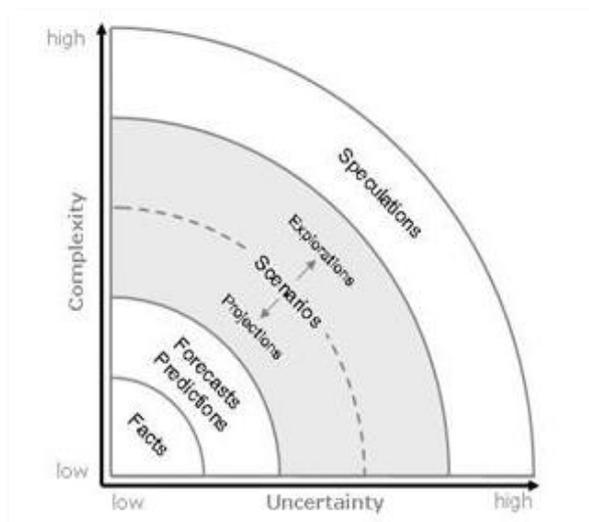
Source: Heinrichs et al., 2010

It is important to emphasise, though, that scenarios should avoid determinism. If depicting the future, it is because of the belief that things can be changed, that there is leeway for policy-making and social participation to actually make a difference.

Inherent to EU and global biodiversity policy is the consideration of different milestones: the current situation, 2020 (EU Biodiversity Strategy targets, Aichi targets) and 2030 (Sustainable Development Goals – Agenda 2030). However, in fact there are two main scenarios involved:

a baseline scenario (based on practice as usual) and a scenario based on actions or responses towards sustainability (conservation policy, changes in sectoral policies, re-design of incentives, implementation of EBM approaches, etc.).

Please note that the definition of the current situation, though, may sometimes be misleading. What is actually needed is a baseline scenario. A baseline scenario is not necessarily equivalent to a scenario (only) describing the current situation but rather the trend if there is no action (towards 2020 and 2030). In other words, it is not what is happening today, which is just part of the story, but rather what would happen if the different drivers exert pressures over European aquatic ecosystems following a specific trend, a pathway from today to 2020 and 2030, which is what is to be assessed.

As a result of that, AQUACROSS will not show the "before vs. after" comparison but rather how sustainable policy as per aquatic ecosystems would move trends away from a baseline scenario (i.e., what would happen otherwise in terms of pressures and impacts – this explains why this is sometimes called "business as usual" or BAU scenario).

It is clear, though, that there are several dimensions that constrain the baseline scenario. These dimensions do not necessarily imply causal assumptions; rather, they are descriptors of the most important attributes of the futures to be analysed. In other words, whilst assessing the different driving factors of pressures over aquatic ecosystems (see Section 2.4), one may have already noticed that the evolution of many of them is contingent on the overall economic outlook (sluggish economic growth, fiscal consolidation policies, debt crises, bailouts, weak domestic consumption, etc.) in a number of EU countries. As such, economic growth, social perception of progress, environmental awareness, etc. are critical dimensions that condition each driver, to a larger or lesser extent, in each country of this project. These dimensions frame the assessment of drivers but cannot be changed at all by conservation policies.

When assessing drivers, some of them may seem somewhat invariant, following a steady evolution over the next 15 years. Others may entail critical uncertainties throughout that same period. Whatever the case, it is crucial to add clarity as per any assumptions made.

It is also critical to bear in mind that any projection is subject to change in its basic assumptions. It is often said that long-term forecasts are always false or that the further into the future you look, the less you see. Both statements seem to hold true but forecasting is nevertheless unavoidable. Be it either explicit or implicit, forecasting is a need. No explicit forecast is implicitly equivalent to accepting the *status quo*. In other words, long-term forecasts might be false but perhaps less false than accepting that nothing would change. In fact, what a baseline scenario portends is that things *will* change, if not necessarily in the desired way. AQUACROSS implicitly emerges from the belief that current trends of drivers and pressures over aquatic ecoxystems are clearly detrimental and unsustainable. In other words, AQUACROSS should be able to convey the transition from 'crisis' to 'vision'.

Although an endless number of foreseeable futures might be explored within AQUACROSS, scenarios are certainly more powerful (and effective in terms of dissemination) when presented as a small set with clear differences.

2.1.2 Building scenarios: making the AQUACROSS Architecture operational

If the general objectives of building a baseline scenario and policy scenarios are to represent the social-ecological interactions and the policy challenges and options in a particular situation or case study, the specific objectives of making the AQUACROSS Architecture (see below) operational and building a baseline scenario are as follows:

▸ Building a shared view of current trends and vulnerabilities as per ecosystems and biodiversity, with a special focus on the economic and institutional failures that must be addressed in the social system and the evaluation of non-linear feedback loops, critical thresholds and the existing risk or hysteresis, as well as irreversible regime shifts.

▸ Integrating diverse disciplines over the wide spectrum of natural and social sciences that have different concepts, definitions, methods, assessment criteria, analytical models and research programmes into a **comprehensive framework** to make the different pieces of knowledge suitable to serve a common social purpose.

▸ **Harmonising and integrating concepts** and metrics across different scales throughout time, space, ecological organisational levels, and policy domains. This improved communication is expected to help overcome knowledge and institutional barriers, to facilitate the identification of new opportunities linked to EBM approaches and to foster the cooperation amongst stakeholders and policy areas that is required to take advantage of synergies and co-benefits associated with the enhancement and protection of ecosystems and their biodiversity across different aquatic realms.

▸ **Representing the outcome of cumulative pressures over biodiversity and ecosystems** as a means to confront stakeholders and make them aware of the consequences of their own decisions. This is expected to result in a much better understanding of impacts over ecosystem structures and functions and of the ensuing detrimental effects on human wellbeing. This comprehensive analysis would contribute to increase the visibility of the opportunity costs of ecosystem degradation and biodiversity decline along with the benefits of their preservation.

▸ Supporting the identification of well-defined **targets** in terms of biodiversity, ecosystem services, functions and structures, and the development of adequate information and decision systems to support their achievement in a cost-effective, efficient and equitable manner (see Section 2.2 on policy objectives below).

▸ Providing a framework to represent and convey uncertainty over scientific knowledge, the foreseeable dynamics of SES and the impact of individual and collective policy responses (see Section 2.6.2 on uncertainty).

These objectives are vast and perhaps not within the reach of existing knowledge even for well-defined case studies as the ones considered in AQUACROSS. The diversity of scientific perspectives involved has led to fragmented pieces of knowledge that limit our ability to understand the relevant social-ecological linkages. As explained in the AQUACROSS concept

(Gómez et al., 2016), stakeholders and governments are doomed to make decisions with incomplete and imperfect information. Yet, the right methodological approach is definitely not about waiting until a better knowledge base is available, but rather about conveying the best available knowledge to the policy arena to generate the positive feedback loops that may improve current environmental responses and drive a new research agenda.

As above, baseline scenarios are not (or not only) representations or predictions of the future. Uncertainties over drivers of ecosystem change and on how social and ecological systems will adapt make the future unpredictable to a very large extent. Nevertheless, scenarios based on data and models are tools to: ascertain the mechanisms and pathways that have led to current policy challenges, to understand the functioning of SES, to synthesise a wide range of information, to assess the effectiveness of policy responses and to provide insights to consider situations relevant for policy-making, such as what the state of biodiversity will be if no action is taken to halt current degradation trends. In other words, what would happen if we insist on traditional policy approaches or if EBM approaches are not sufficiently implemented?

While data and science are the basis for understanding particular layers and processes of this analysis, the effective uptake of this knowledge by stakeholders requires finding suitable communication tools able to factor in the stylised facts provided by science into meaningful narratives and storylines. These narratives are an integral part of the baseline scenario as they allow the integration of the different processes taking place at the SES into a common and eventually shared representation of policy problems.

Box 2: Examples of scenarios for reaching biodiversity targets

a) **Examples of the use of scenarios and models in agenda setting, policy design and policy implementation relating to the achievement of biodiversity targets across a range of spatial scales**. The diagram indicates the typical relationships between spatial scale (top arrows), type of science–policy interface (upper set of arrows at bottom), phase of the policy cycle (middle set of arrows at bottom) and type of scenarios used (lower set of arrows at bottom):
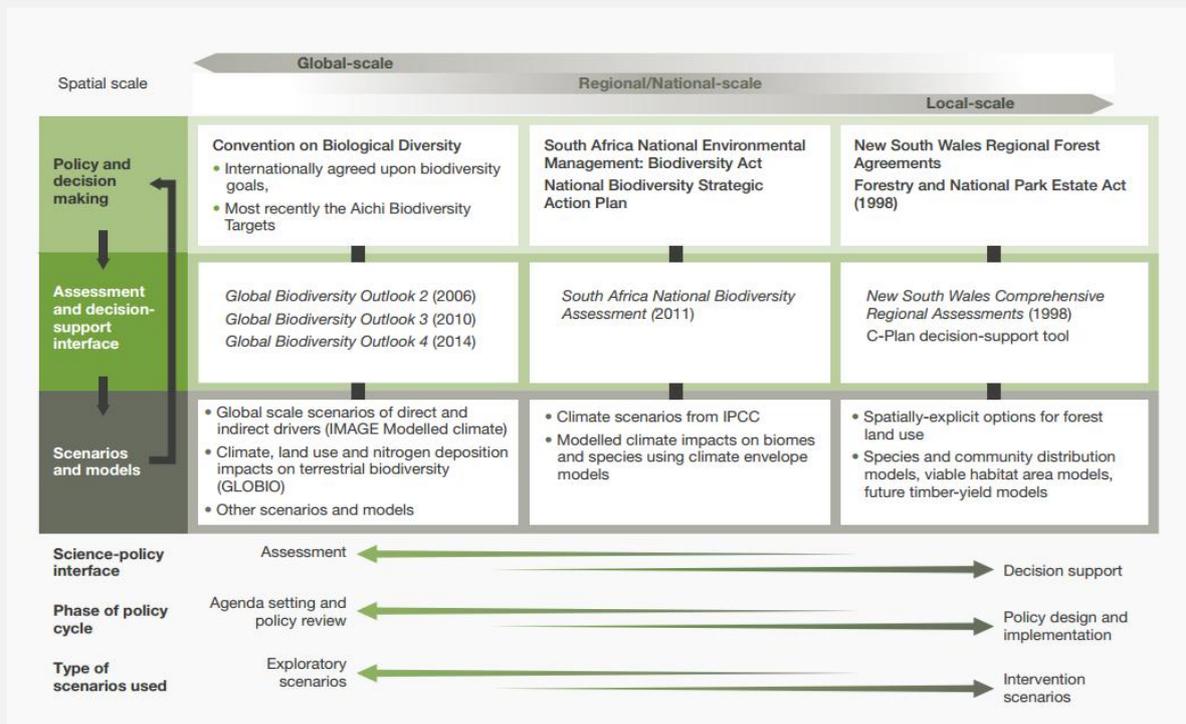
b) Influence of human use on biodiversity: **the present situation vs the target situation** under WFD and HD:
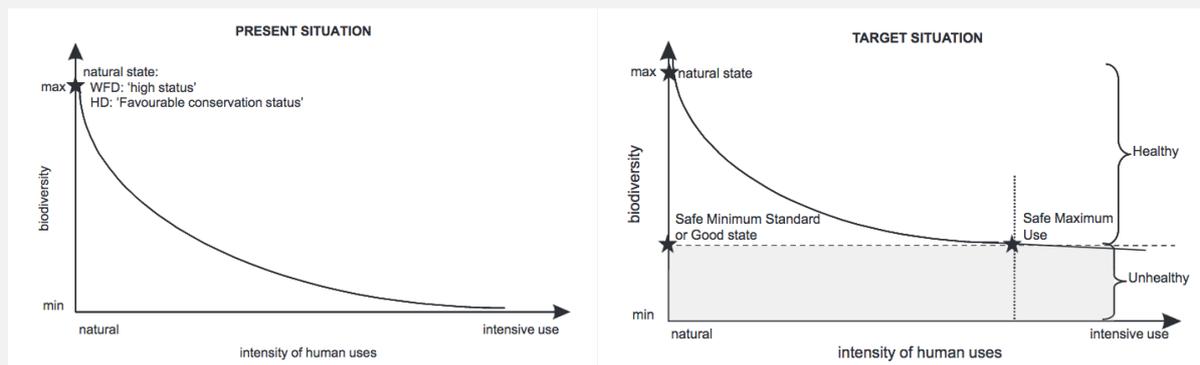
Source: a) IPBES, 2016, p. 21; b) Schneiders et al., 2011

2.1.3 Co-building baseline and policy scenarios to progress towards ecosystem-based management

As above, the AQUACROSS EBM approach is to be primarily driven by policy objectives (see Section 2.2). Yet, overcoming current practice and advancing towards the implementation of holistic and integrated EBM responses is very demanding in terms of science and policy (see Section 1.3.3).

The holistic approach adopted in AQUACROSS requires building comprehensive and complex scenarios able to represent the problem (as in the baseline scenario) and the alternative potential solutions (as in the management scenario). As a result of this holistic nature and complexities entailed in this assessment, there is neither a 'universal' EBM approach nor only one EBM implementation path to such complex problems, and adding scientific knowledge may well not necessarily close prevailing knowledge gaps (e.g., Espinosa-Romero et al., 2011; Dankel et al., 2012).

Therefore, to guide this process we recommend a consideration of the "interaction triangle" (Röckmann et al., 2015) representing the interaction pathways between science decision-makers and other stakeholder groups (Figure 2).

Figure 2: 'Interaction triangle'
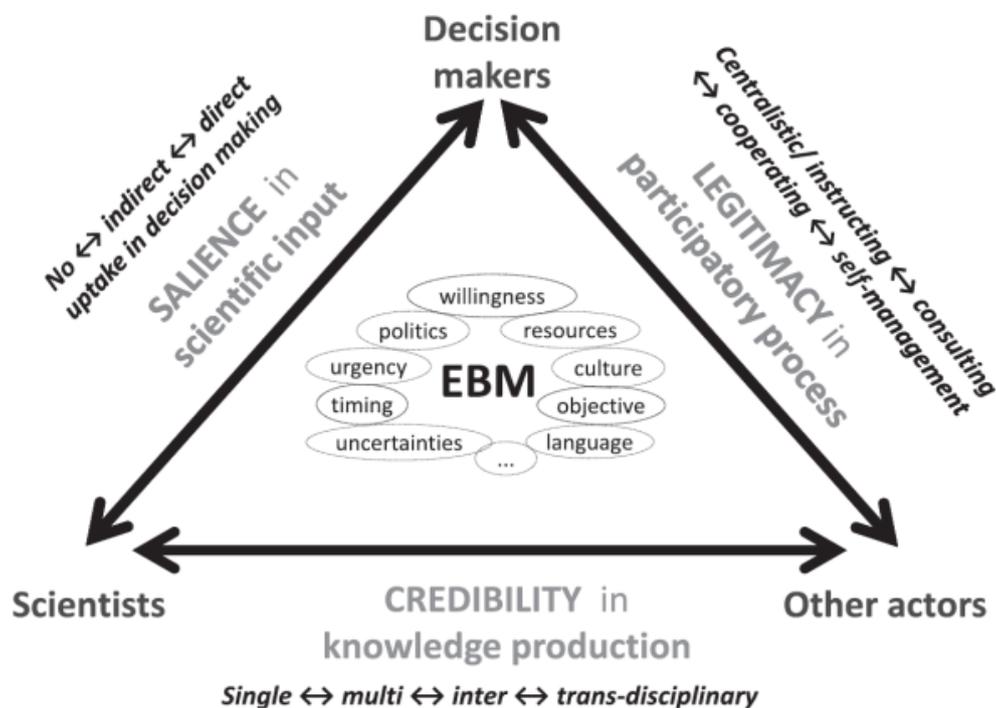
Source: Röckmann et al., op. cit.

Within AQUACROSS, EBM is supposed to be grounded on the best 'knowledge base' (FAO, 1995; Tallis et al., 2010; Fanning et al., 2011). As represented in the lower left corner of the above diagram the role of science is double-edged. On one side it is intended to be relevant to decision-making; on the other it is essential for the credibility of social knowledge and for the legitimacy of the policy decisions it intends to inform and improve. These interactions are handy to develop the three main distinctive features of the science-policy nexus to move towards the implementation of EBM:

- ▸ EBM involves a deliberate strategy to increase the salience of science through its direct contribution to improve policy-making processes. The role of scientific knowledge is determined by what science can offer to policy and by the interaction with policy-makers. Three situations can be identified regarding the relative and progressive advance of science and its associated potential to inform policy-making:

  - ○ No policy relevance of science: this may occur when the scientific state-of-the-art might be too new, too preliminary or too uncertain to be directly applied. This may also be the outcome of a strong top-down political process when the practicality of science is not fully appreciated. For example, when decision-makers are under extreme time pressure to make an urgent decision, thus not being able to wait for up-to-date scientific inputs.

  - ○ Indirect relevance and interaction with policy: this refers to contexts where scientific discovery, while intrinsically relevant, does not directly provide the means for its application to management. For instance, while innovative and insightful, early environmental risk assessments were unlikely to lead to immediate and direct policy change (Mitchell et al., 2006: 309-310). Other cases occur when discovery and new scientific evidence point towards areas that call for new management actions different from current practice. A time lapse can also apply the other way around, i.e. scientists becoming interested in and more capable of studying a particular issue, as a reaction to management demands (Mitchell et al., 2006).

  - ○ Direct relevance of science for policy: this occurs when the science-policy interaction increases the potential for scientific output to be directly applied in decision-making. The framing of problems, the co-building of baseline and policy scenarios from the onset is crucial to define an applied research question (Röckmann et al., 2012). Mitchell et al. (2006) propose lessons that can aid in bridging the gap between scientists and decision-makers and making scientific input more salient, e.g.: focus on processes and not only (scientific) outputs, acknowledging decision-makers' concerns, perspectives and values, involving other actors, and making use of existing networks.

- ▸ Under an EBM approach, science is not only intended to inform and to make technically sound decisions but rather act as a means to build a reliable knowledge base through interaction between scientists and stakeholders.

  As explained in the AQUACROSS innovative concept, the holistic and complex nature of EBM requires different approaches of knowledge production and this entails both

quantitative and qualitative approaches. Quantitative methods are desirable and generally required, to give precision and to appreciate the breadth and magnitude of uncertainties involved both in knowledge and policy. Nevertheless, qualitative approaches are essential for scoping and framing, such as determining model boundaries, setting assumptions, interpreting results, but also for generating in-depth knowledge about the effects of multiple social, political and economic factors.

In addition to research-based knowledge, traditional community knowledge on ecosystem management (Berkes et al., 2000; Huntington, 2000; Anadón et al., 2009) is increasingly considered as useful in marine ecosystem management to deal with uncertainty, offering "a means to improve research and also to improve resource management..." (Huntington 2000:1270). Yet, such evidence-based knowledge cannot be easily analysed, compared, or linked to information on a broader scale (Wilson, 2009). The spectrum of approaches to knowledge production ranges from single, via multi- and inter- to transdisciplinary.

Regarding the management of complex environmental problems, Haapasaari et al. (2012a:1) conclude that the "scientific knowledge base has to be expanded in a more holistic direction by incorporating social and economic issues" in addition to the natural science basis. A review on marine and coastal research "argues that theories and methods should conform to a perspective that ocean management is a societal activity with diverse goals ideally informed by interdisciplinary information" (Christie, 2011:172). A 'novel' EBM approach therefore increasingly (but not always) requires a move from single- towards inter- and trans-disciplinary approaches (Haapasaari et al., 2012a; Phillipson and Symes, 2012).

▸ The EBM approach promoted by AQUACROSS considers the science-policy link of upmost importance for the legitimacy in the participatory processes. The co-building of baselines and policy scenarios, and the full policy-making process, through increasingly better informed stakeholder involvement can strengthen democratic cultures and processes (Webler and Renn, 1995), bring additional knowledge and values into decision-making in order to make better decisions (Badalamenti et al., 2000; Renn, 2008), provide greater legitimacy (Raakjaer and Vedsmand, 1995; Raakjaer and Mathiesen, 2003), increase trust (Renn and Levine, 1991; Munton, 2003; Luoma and Löfstedt, 2007; de Vos and Mol, 2010; Young et al., 2013), enhance compliance (Jentoft, 2000; Christie et al., 2009; Christie, 2011), and reduce the intensity of conflict (Young et al., 2013).

A knowledge based decision-making process can result in increased management efficiency, equity, sustainability, reduction of administration and enforcement costs (Raakjaer and Vedsmand, 1995), making management not only more legitimate, salient or credible, but also enforceable and realistic (Fiorino, 1990; van der Sluijs, 2002; Craye et al., 2005; Leslie and McLeod, 2007; Renn, 2008; Wilson, 2009; Tallis et al., 2010; de Vos and van Tatenhove, 2011; van der Sluijs, 2012). A gamut of interaction between decision-makers and stakeholders was described as "gradations of citizen participation" identifying a "typology of eight levels of participation" (Arnstein, 1969:217). The bottom

rungs describe levels of "non-participation", meaning "not to enable people to participate in planning or conducting programs, but to enable power holders to 'educate' or 'cure' the participants".

A bit higher up in the ladder, the levels of informing and consultation allow participants "to hear and to have a voice"; however, participants "lack the power to ensure that their views will be heeded". The top levels of the ladder "are levels of citizen power with increasing degrees of decision-making clout". This continuum thus assigns different roles and responsibilities to the managers and to those being managed. Neither top-down government centralistic management, nor bottom-up self-management is necessarily the best way for natural resources management. The important aspect is to be transparent about the roles and responsibilities expected from the involved parties. Many have highlighted the importance of early involvement of stakeholders, i.e. in the problem framing/ scoping phase of a participatory process (Dreyer and Renn, 2011; Haapasaari et al., 2012b; Röckmann et al., 2012). Stakeholders' roles in the process should be clarified (Ferreyra and Beard 2007; Mostert et al., 2007; Young et al., 2013), and "a common vision including the objectives for marine EBM" should be defined (Leslie and McLeod, 2007:542; Fanning et al., 2011). Clarity and transparency can help to prevent misunderstanding.

2.1.4   The baseline scenario: representing the social-ecological system as a whole

The AQUACROSS Architecture stands for the methodological approach to build up a baseline scenario. That is to say, the whole heuristics that allows us to integrate and synthesise scientific knowledge in a fashion that is familiar to stakeholders and managers and that is suitable to inform EBM approaches to jointly manage complex SES.

These Heuristics of the project (see below) are oriented towards management. Beyond taking stock of existing knowledge and representing the state-of-the-art, it aims at mobilising scientific knowledge to improve social capacities in order to provide better responses to ecosystems and biodiversity management challenges.

In practical terms, the AQUACROSS Architecture aims at mobilising knowledge to (i), confront stakeholders and institutions with the outcomes of their current decisions and, (ii) support collective decision-making to integrally manage ecosystems by comparing and assessing alternative courses of action.

Along these lines, the main methodological challenge to realise the first general objective consists in making a holistic approach truly operational through the identification, effective design and successful implementation of EBM approaches to respond to the challenges of biodiversity across freshwater, coastal and marine ecosystems.

Figure 3: The AQUACROSS Architecture
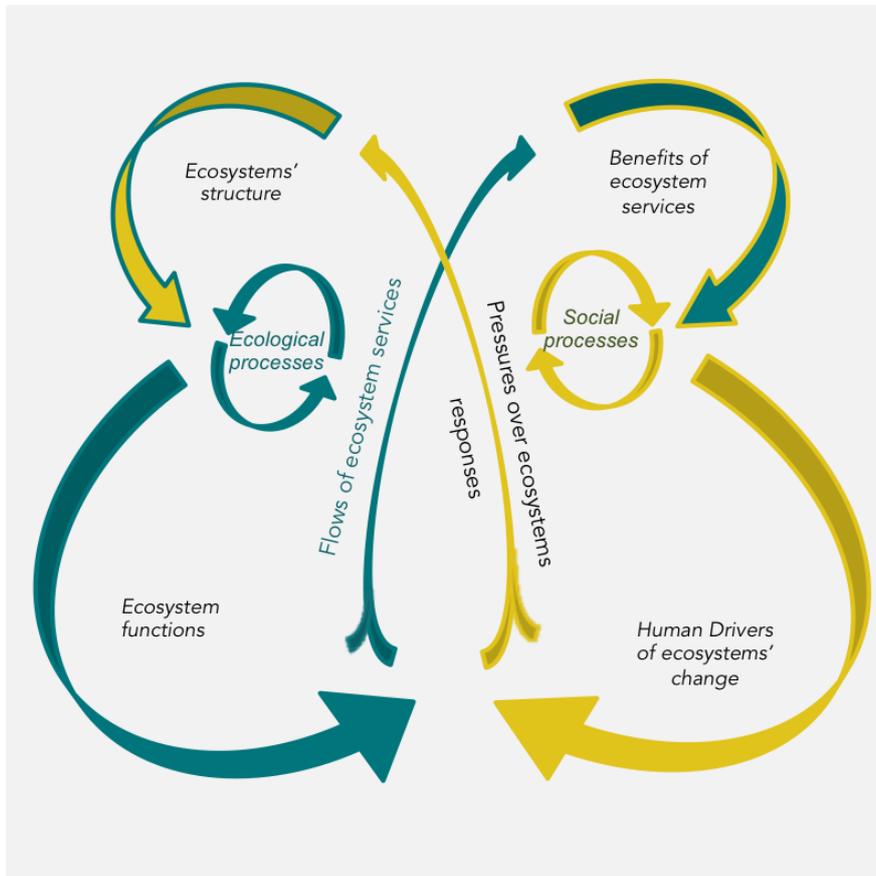
Source: Gómez et al., 2016

From an analytical point of view the whole AQUACROSS Architecture can be disentangled into two separated but closely related sets of linkages or interactions:

▸ On the one hand, we can analyse the detrimental consequences over the ecosystem that result from the satisfaction of multiple demands of services provided by nature to society. This is the demand side of the AQUACROSS Architecture, represented by yellow arrows in Figure 3 and showing how the demand and use of naturally provided services is an outcome of social processes, including markets and governing institutions, and determined by multiple factors (such as population and economic growth, climate change, technology development, etc.). These demands of services such as freshwater, minerals, fish biomass, water security or pollution control, etc., and the way they are met (through water impoundment, trawling, deep sea mining, dredging, drilling, etc.) result in pressures over ecosystems and further changes in their structure. This demand side is explored in Section 2.4 of this document.

▸ On the other hand, we can analyse the potential of ecosystems to continue providing ecosystems services on which human life, the social system and the ecological system

itself depend, and how all this affects human wellbeing. This is the supply side of the AQUACROSS Architecture, represented by green arrows linking ecological and social processes. This analysis allows understanding the functioning of ecosystems and how changes, induced by human actions, are linked to human welfare and sustainability. This supply side is explored in Section 2.5 of this document.

The supply and the demand sides of the analysis are linked to one another through complex adaptive processes taking place in the social and the biophysical systems (see Figure 3). On the social side, these processes include climate and land use change adaptation, institutional development, technical innovations and other social processes that are increasingly shaped by contemporary environmental challenges. On the ecological side this includes adaptive processes such as biodiversity depletion, non-indigenous and invasive species, ecological tipping points, etc. and other changes increasingly moulded by the influence of social decisions.

Box 3: Lessons from scenarios on invasive species on islands

Understanding the system:

Islands and their respective terrestrial and aquatic ecosystems are highly diverse and host approximately 20% of all known species. Given their small size and fragile ecosystems, islands and their unique biodiversity have been disproportionately impacted by invasive species with consequent environmental, economic and social effects.

Invasive species can have practically irreversible consequences by changing abiotic and biotic factors within ecosystems and plant/animal communities. Multiple invasive species in a system can increase the complexity of management efforts and may facilitate trophic cascades that fundamentally alter ecosystem structure and functioning.

Uncertainty around rates of introduction, establishment, inter-species interactions and a range of climate scenarios present difficulties in forecasting the full complexity of invasive species potential impacts on island ecosystems. The tipping point for invasive species' impacts on islands is likely at the stage shifting from their establishment to spread, whereas the critical point for managing impacts is preventing the introduction in the first instance (i.e., pre-border or at border quarantine) or eradicating them soon after their introduction.

Setting policy priorities and opportunities:

Given their isolation and size, islands can provide an ideal environment for the development and application of biosecurity measures and management techniques. The majority of successful invasive species eradications have been on islands.

Efforts to prevent introductions and manage the spread of invasive species are inherently exercises in uncertainty, which can benefit from improved modelling, use of risk assessment and better data. Support for biosecurity policies and invasive species management in islands is likely a sound investment for protecting unique and abundant biodiversity and key ESS.

Source: Leadley et al., 2010

2.1.5   Framing the decision context

The second general objective consists in developing a management framework by applying the above-mentioned Architecture so as to fulfil the following decision-support objectives:

▸ Framing management challenges (such as decline in biodiversity and fish populations) within precise ecological (geographical area, relevant ecological processes, etc.) and institutional boundaries (stakeholders, regulations in place, property rights, development trends, etc.).

▸ Identifying and agreeing on management objectives, considering primary EBM objectives as well as ulterior objectives within the SES (see Section 2.2 below).

▸ Identifying opportunities and barriers linked to alternative ways to pursue management objectives (such as synergies among policy domains, opportunities linked to reinforced ecological processes, barriers linked to crowding out or rebound effects, co-benefits, forward and backward linkages, etc.).

▸ Evaluating gaps and deficits in the ecosystems' structures and functions as well as in social institutions and capacities that need to be bridged in order to pave the way for the feasibility of management objectives.

▸ Assessing available alternatives to cope with management challenges in terms of cost-effectiveness, cost-benefit analysis, multi-criteria decision and other relevant methodologies to assess policy alternatives with effectiveness, efficiency, fairness, legitimacy and other socially and environmentally relevant criteria.

▸ Developing management-oriented indicators to support the assessment of challenges, objectives, policy options, etc. and guaranteeing the standardisation of definitions and metrics to make both the assessment and comparisons relevant for management.

▸ Conveying evidence-based information relevant to policy-making in such a way that can be understood and used by stakeholders to screen out policy alternatives and understand the foreseeable consequences of the different courses of action (including business as usual and management scenarios).

▸ Supporting the construction of a shared understanding of foreseeable consequences and the uncertainties linked to the different management alternatives as well as reinforcing collective decision-making in the face of uncertain outcomes.

Framing the policy question requires looking at all components of the SES. Nevertheless, we will focus primarily on ongoing processes that result in detrimental consequences for biodiversity and aquatic ecosystems, hence having a negative impact over ecosystems and the services people get from them. Along these lines socially-relevant ecological processes are the primary concern of AQUACROSS. Thus the first step to set the policy problem consists in defining the degradation processes taking place in the ecological system. Defining a policy challenge within this context is equivalent to describing and representing that degradation taking place in the ecosystem (i.e., proliferation of invasive species, overfishing, water

scarcity, exposure to floods, etc.) but also defining why this problem is relevant for human welfare (representing, for example, foregone benefits, welfare losses due to a lower provision of critical ecosystems services, increased risk of a regime shift after an extreme event, etc.).

The primary focus on environmental problems does not mean that the social system is not relevant to define the policy challenge, or that actions must be limited to restore the environment. In fact, causes of ecological degradation are to be found in the social system (as those ecological degradation processes are driven by market and/or institutional failures) and its reform is a prerequisite to restore the sustainability of the whole SES.

Explaining ecological problems is equivalent to elucidating the social drivers of ecosystem change. Also how these drivers are translated into pressures over ecosystems, result in changes in the way these systems function, particularly in their potential to continue providing ESS, which clearly has impacts over human welfare triggering responses that sometimes are not adequate.

Thus reshaping the social system in order to be able to respond to the sustainability challenge is an integral part of the policy response. This reform is inspired by the need to build institutional capabilities to take advantage of the opportunities linked to implementing EBM approaches (such as cooperation to take advantage of synergies and sorting trade-offs out, etc.) so as to enhance the sustainability of the whole system.

2.1.6 The baseline scenario: where to look at to identify policy challenges

One relevant working hypothesis of AQUACROSS is that prevailing best practice consists in optimising the delivery of particular ESS (food, water, energy, safety, etc.) and seeks to maximise the production of specific components of the system (such as water quantity or fish biomass), through controlling others (water storage, flood risk, etc.), at a limited scale (mostly local), and for a limited time frame (mostly in the short term). This practice sets aside or assumes no changes in the functions and structure of ecosystems on broader spatial scales and through the medium and long term (Walker and Salt, 2006; Levin, and al., 2013). This basic idea provides the key for searching both management problems and opportunities for improvement.

A second important hypothesis is that economic (and decision-support) models that do not consider complex adaptive natural and social systems may well lead to socially and environmentally undesirable outcomes. Despite what optimal resources management models may suggest, dynamic systems cannot stay steadily in an ideal optimal status chosen to deliver maximum sustainable yields of fish, freshwater or wood, just to mention a few examples. Furthermore, ecosystems and natural resources are by any means not only affected by single disturbances, such as extraction rates or pollution loads, but rather by disturbance regimes represented by the pattern and dynamics of disturbances that shape the ecosystem itself in the long term (see Pickett and White, 1985).

Although minor changes in complex systems are often linear and incremental, ecosystems are only stable within critical thresholds and might change into alternative stable states due to disturbance regimes (Beisner et al., 2003). Permanent disturbances and extreme events such as droughts or floods, storm surges, etc. are able to reorganise system properties and affect biodiversity. For instance, different studies of aquatic, forest and other ecosystems show that smooth changes can trigger sudden variations in regimes and lead to the irreversible loss of ESS (Scheffer et al., 2001). Similarly, minor changes in sediment transport may trigger a catastrophic drift of stream invertebrates (Gibbins et al., 2007). Surprises, such as silting, dead zones in river mouths, fisheries collapse, etc. make visible the drawbacks of traditional approaches and prompt for alternative methodological approaches such as the one proposed in AQUACROSS. Therefore, identifying slow variables, or on-going degradation processes such as sedimentation, siltation, invasive species colonisation, water scarcity, etc. and asking for the possibility of regime shifts in the face of extreme events is in many cases an important step to define the policy challenges at stake.

Box 4: A working example: Stepwise application of the AQUACROSS concept to explore major linkages, identify research needs and policy challenges
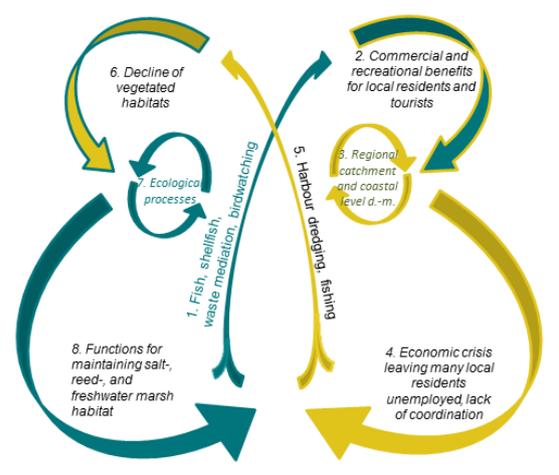
1-2. The lagoon area from the Vouga river supports multiple services, some of them uniquely connected to the special saltmarsh habitat. Commercial and recreational fisheries traditionally contribute to local residents well-being; however, the economic crisis emphasised these activities' importance for complementing staple food beyond the legal quotas. Furthermore, the different marsh types contribute to mediation of waste and toxics from inland freshwaters and regulate a unique habitat suitable for a variety of birds also attracting tourists.

3. At regional level, participatory meetings are held to present current state and plan of activities for the catchment area and lagoon (river basin management plans, WFD) and at the coastal level (coastal zone spatial planning and seashore protection), which are discussed by different stakeholders and the general public (that wishes to participate in the meetings). Examples of conflicts are recreational fishing and hunting within touristic activities, both in the role of provisioning and cultural services.

4. Multiple drivers affect the region, both from outside, such as the economic crisis, but also from inside, where a lack of coordination between different interest groups, such as among the harbour business and fishing activities, leaves several conflicts unresolved.

5. That is why sectoral interests were pushed for in the past with, for instance, dredging activities in the harbour region, which heavily affects the hydrological system of the lagoon area.

6. Dredging changes velocity of water in the lagoon together with an increase of salinity and reduces marsh habitats.

7-8. It is yet to be investigated how the saltmarsh habitat is expected to change under the ongoing pressures and how this translates into important functions, such as nursery areas for commercially valuable fish.

2.1.7 Integration of resilience principles in building the baseline

To disentangle the complexity of reaching multiple biodiversity targets and ensuring a diverse set of ESS in SES, resilience thinking supports the construction process of the baseline. Seven generic principles were described for enhancing the resilience of ESS, i.e. the capacity of a SES to sustain a desired set of ESS in the face of disturbance and ongoing change (Schlüter et al., 2015). Two particularly important principles for building the baseline for a specific case according to the AQUACROSS Architecture are complex adaptive systems thinking (P4-CAS) and slow variables and feedbacks (P3) (see Schlüter et al., op. cit. for an overvirew of the 7 principles).

CAS are made up of many interacting components that are individually and collectively adapting to change, enabling them to self-organise and evolve, often yielding emergent properties at different scales (Norberg and Cumming, 2008). Complex adaptive systems may shift between alternative regimes, often abruptly and irreversibly (Scheffer et al., 2001). Identifying suitable system boundaries that integrate not only the main interacting components but also the leverage points for management to induce a desired change for improving biodiversity targets and ESS provision is crucial to design the baseline.

Feedbacks occur when a change in a particular variable, process or signal leads to changes in other variables that eventually loop back to affect the original variable, process or signal. Slow variables are variables that change on time scales that are much slower than conventional time scales. Slow variables can mask feedbacks while driving the system towards a threshold (e.g., phosphorous accumulation in lake sediments). Since these processes act at multiple scales, it is relevant for the baseline to identify them before entering the individual linkages of the AQUACROSS architecture.

A challenge in applying those principles for a particular case lies in their interrelation since any of the principles in isolation rarely leads to an overall, enhanced resilience of ESS. However, three key mechanisms of their interrelation could be identified (see Schlüter et al., 2015):

1 "Increasing understanding of critical SES (social-ecological system) components and processes (P3-Slow variables & feedbacks, P4-CAS thinking, P5-Learning) as well as suitable management options (P5- Learning, P6-Participation, P7-Polycentricity).

2 Preparing the SES for unexpected events by creating awareness of their likelihood (P4-CAS thinking), and providing alternative approaches and ways of dealing with emergent issues when suddenly needed (P1-Diversity, P7-Polycentricity), and

3 Enhancing response capacity by providing a diversity of response options (P1-Diversity), building the trust needed to make decisions and take action (P6-Participation), and providing ways to make use of different responses at the right scale (P2-Connectivity, P5 – Participation, P6-Learning, P7-Polycentricity)."

Complex interactions in SES make it challenging to isolate a particular system property or principle (e.g., diversity) and to establish its connection to the resilience of ESS. Unravelling

the different principles is more of an analytical construct than a reflection of individual, discrete factors operating within a SES. Even if the effect of a particular principle is known, the fact that SES continually evolve and change over time implies that these causal links may also change (Schlüter et al., 2015). At a practical level, the relevant system processes often happen over long timescales, which make it difficult to assess the effect of a principle within the time frame of a typical empirical study or management experiment. Furthermore, the indicators needed to monitor long-term, nonlinear, and variable change are generally not well developed and in some cases may require non-traditional methods and ways of thinking in their assessment (Moss et al., 2010; Halpern and Fujita, 2013).

It is of paramount importance to note that none of the principles are either necessary or sufficient, or a panacea for environmental governance. Applying these principles involves viewing them not as end goals but rather as processes or mechanisms to create the right conditions that allow for resolving collective action problems associated with multiple trade-offs (Schlüter et al., 2015). One approach to unveil those trade-offs and identify useful resilience principles for improving overall ESS governance is to look into how particular services are co-produced and interact with each other.

Management strategy evaluation (MSE) was first conceived as part of (marine) fisheries management and involves simulation to compare the relative effectiveness for achieving management objectives of different combinations of data collection schemes, methods of analysis and subsequent processes leading to management actions. MSE can be used to identify a 'best' management strategy among a set of candidate strategies, or to determine how well an existing strategy performs (Punt et al., 2016). MSE (i.e., the evaluation of management strategies using simulation) is widely considered to be the most appropriate way to evaluate the trade-offs achieved by alternative management strategies and to assess the consequences of uncertainty for achieving management goals. MSE overcomes many of the concerns with any of the traditional approaches, including that the full range of uncertainty can be taken into account and decision-makers may consider longer-term trade-offs among management objectives, instead of focusing on short-term considerations only. As this links to several of the EBM principles, MSE can probably be considered the most appropriate approach to assess the effectiveness of management. While MSE was mostly applied as part of fisheries management it is potentially applicable to any type of resource management and the same best practices apply (see Punt et al., 2016).

2.1.8 Science focus: models and tools for a stepwise building of baseline and EBM policy scenarios of biodiversity

This section describes a full workflow that can, as a whole or parts of which, be applied to help operationalise EBM in the different case studies. Figure 4 visualises the individual parts of the workflow (a/A to i). Subsections that deal with scenario building, modelling, projections, and prioritisations of biodiversity and ESS are elaborated below. Statistical relationships among species occurrence data from monitoring campaigns and respective information for environmental variables are established (Figure 4a). This relationship is used

to model the current species distribution at time (t) = 0. Whether statistical, mechanistic, or hybrid models are used to simulate the distribution of a species across the landscape, the input data generally consists of the two major data types: species' geographic occurrence data and environmental variables to be associated with their occurrence (Elith and Leathwick, 2009).

Over the past decades, there has been a massive increase in publicly available Geographic Information System (GIS) data, regarding both species occurrence data and environmental data. Although species distribution models as such are not new (see point-based models in e.g., RIVPACS that predict the occurrence of a species at a specific location: Moss et al., 1987), the availability of data over large spatial and temporal extents has pushed the field towards being a 'standard' assessment tool of the species – environment relationship in ecology and biogeography. Originally developed in the terrestrial realm SDMs have been successfully adopted for freshwater and marine ecosystems (Robinson et al., 2011; Domisch et al., 2015a).

The choice of the modelling technique – statistical, mechanistic, or hybrid – mainly depends on data availability, time frame, and effort of a given study. On the one hand, statistical models such as correlative species distribution models, are less data hungry than the mechanistic and hybrid counterparts which can often only be accomplished for single species or a limited set of species due to the lack of data (e.g., physiological tolerances and dynamics, biological interactions), and knowledge of the species' autecological preferences (Morin & Thuiller, 2009). On the other hand, input data in statistical models has to be selected very carefully to maintain causality between species and environmental data (Dormann et al., 2013). When applied carefully, correlative SDMs have the potential to highlight important species–environmental relationships and in case of the application of scenarios to show trends in species and community range changes and turnover over large spatial extents (the latter, see below; Elith and Leathwick, 2009).

a. Species occurrence data

Species occurrence data can be derived from ad-hoc observations or systematic surveys, and indicate which species was found at a given geographic location and time.[2]

For instance, Domisch et al., (2015a) discussed the key issues to build SDMs in stream ecosystems, while Snickars et al., (2014) analysed them from 145 peer-reviewed field-studies, focused on interaction among predictors and regional effects in coastal waters. Mesa et al. (2015) suggested predictive models in marine ecosystems (abundance vs presence/absence) based on catches, with conclusions on monitoring and research programmes.

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[2] Data sources for freshwater species occurrences are the Global Biodiversity Information Facility, the Freshwater Information Platform, fishnet2, country-based monitoring programmes, e.g. according to the European Union Water Framework Directive (WFD; 2000/60/EC), and previous EU projects and collaborations such as WISER, STAR, EFI+ etc. Likewise, species occurrence data in the marine realm is available in GBIF, Ocean Biogeographic Information System, ReefBase and additional long-term monitoring programs from local research institutions.

Figure 4: Workflow for evaluating alternatives and prioritising conservation and ecosystem services (ESS) delivery areas for the application in the AQUACROSS case studies
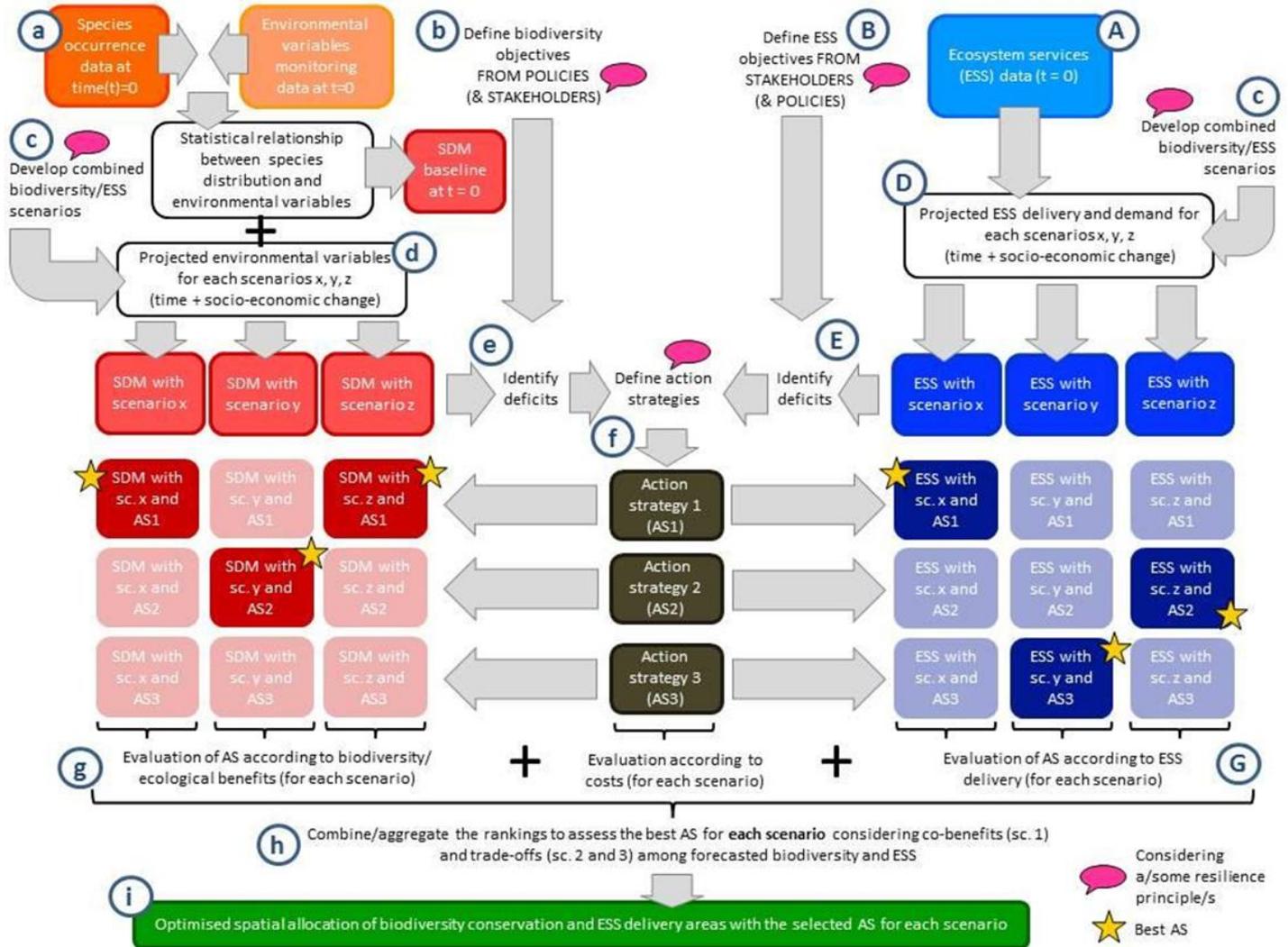
SDM = Species Distribution Model. AS = Action Strategy

Source: Own elaboration (Simone Langhans, FVB-IGB)

b. Environmental data

Compared to the terrestrial realm, the development and availability of high-resolution and range-wide environmental data in the freshwater and marine realms has been more recent. On the one hand, the difficulty lies in translating remotely sensed data into meaningful variables for the specific realm. In freshwaters, for instance, it is crucial to account for the stream network and the upstream connectivity, opposed to a specific land use type at one location as done in the terrestrial realm (e.g., amount of upstream agricultural area influencing the downstream river reaches (Peterson et al., 2013; Kuemmerlen et al., 2014). Likewise in marine SDMs, the connectivity needs to be maintained by accounting for ocean

currents that are vital to be depicted correctly. Moreover, marine organisms occur in a three-dimensional space given by the water column. Hence, environmental variables measured at the surface may differ from those measured in the water column or at the benthos (Dambach and Rödder, 2011). In essence, the freshwater and marine SDMs need to be spatially explicit, with additional levels of complexity compared to those applied in the terrestrial realm (Robinson et al., 2011; Domisch et al., 2015a).

c. Sources of environmental data

To date, there are several global datasets for range-wide environmental data. Regarding freshwaters, a variety of environmental data is available, such as the gridded 1km freshwater data for rivers and lakes from Domisch et al., 2015b[3] that takes into account the upstream connectivity, data based on catchment polygons[4], environmental characteristics for lakes[5], wetlands[6], reservoirs and dams[7] and inland water bodies on a 30m spatial grain[8]. Moreover, continental datasets can provide additional information at a more detailed level, such as the European Catchment Characterisation and Modelling (CCM) database[9] (de Jager and Vogt, 2010). Marine datasets with a global and range-wide coverage exist in various spatial and temporal coverages and data types, such as temperature, chlorophyll, nutrients, salinity, chemistry, and bathymetry. These data are available from the Bio-Oracle dataset[10], the MARSPEC dataset[11], and the CMIP5 database[12]. Moreover, high-resolution bathymetry and sea floor topography is available from the SRTM30_PLUS dataset at approx. 1km spatial grain dataset[13]. To obtain a vertical stack and, hence, a 3D structure of the marine environment, tidal changes on coastal regions, custom oceanographic models and extrapolations need to be considered and included as done, for instance, for the North Sea (Reiss et al., 2011).

There are, however, challenges in data collection, such as species-level biases in occurrence data (Meyer et al., 2016 and references therein) and knowledge gaps and deficiencies in marine ecosystems data, that complicate the operational approach (She et al., 2016) also for river restoration (Downs et al., 2011), or those related to resolution and scale in coastal study cases (Stanev et al., 2016). Rose et al., (2010) discussed end-to-end models that can deal with bottom-up and top-down controls that operate simultaneously, vary in time and space and are capable of handling the multiple impacts expected under climate change in marine ecosystems.

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[3] www.earthenv.org/streams

[4] www.hydrosheds.org: Lehner and Grill, 2013

[5] www.hydrosheds.org/page/hydrolakes: Messager et al. in review

[6] www.worldwildlife.org/pages/global-lakes-and-wetlands-database: Lehner and Doll, 2004

[7] http://atlas.gwsp.org/: Lehner et al., 2011

[8] www.landcover.org/: Feng et al., 2016

[9] http://ccm.jrc.ec.europa.eu/php/index.php?action=view&id=23

[10] approx. 9km spatial grain, www.oracle.ugent.be/: Tyberghein et al., 2012

[11] approx. 1km spatial grain, www.marspec.org: Sbrocco and Barber, 2013

[12] approx. 150 to 300km spatial grain, https://pcmdi.llnl.gov/projects/cmip5/: Taylor et al., 2012

[13] http://topex.ucsd.edu/WWW_html/srtm30_plus.html: Becker et al., 2009

d. Species distribution models (SDMs)

Regarding the SDM itself, the methods used across the realms are very similar. SDMs build upon a statistical relationship based on species and environmental data. In a first step, the species presences (and absences, if available) are analysed in an n-dimensional environmental space defining the habitat affinity given the input variables. In a second step, this species-habitat relationship can be mapped and projected across the study area, yielding a probabilistic habitat suitability map (Elith and Leathwick, 2009). For further analyses, binary presence or absence of a species may be required that can be calculated using a threshold or cut-off value based on the specific model evaluation and discrimination ability (Allouche et al., 2006).

The choice of the statistical modelling algorithm and method ranges from single algorithms (e.g., MaxEnt), (Phillips and Dudik, 2008) to using an ensemble-modelling framework (Araujo and New, 2007), in which multiple algorithms from different classes are combined into a consensus prediction (e.g., regression, classification tree, and machine learning methods (Thuiller et al., 2009)). Beside the quality of the input data, one of the major sources of relative uncertainty has been identified as stemming from the modelling algorithms themselves (Diniz-Filho et al., 2009). Hence, ensemble models are able to show the trend among various techniques and are, therefore, considered more robust compared to single algorithms. Regardless of the choice, the specific settings in each algorithm have to be selected carefully (see e.g., Merow et al., 2013, regarding MaxEnt).

Bayesian techniques are starting to gain more attention in the SDM community because of their flexibility in terms of data input and the ability to depict model-based uncertainties given the available data (e.g., Latimer et al., 2006). Though such models still require high computational power and time, recent advances (such as the hSDM-package in R, Vieilledent et al., 2014) have been promising in terms of application and measures of uncertainty in the species-environment relationship and, consequently, highlighting areas of uncertain predictions as probabilistic distribution maps (Domisch et al., 2016; Wilson and Jetz, 2016).

e. Scales and uncertainty

There are several sources of uncertainty in SDMs that need to be addressed for each species and study area individually. For the species data, this includes accounting for a geographic sampling bias (e.g., political borders, easy-to-access sites are visited more frequently), changing sampling campaigns and schemes over time, and a varying detectability of species (Lahoz-Monfort et al., 2013). In addition, the spatial and temporal scale of the species data has to match those environmental data. For instance, species level data can be aggregated to coarser grains (e.g., from point location to a drainage basin), or temporal scales (e.g., pooling monthly sampling schemes to a multi-year dataset). However, the cost of such procedures is a coarsening of the species-environment relationship, which introduces additional uncertainties in model outputs (Lauzeral et al., 2013; Domisch et al., 2015a). For a further discussion on scales and uncertainties please refer to Section 2.6.

f. Setting policy objectives

Biodiversity objectives are defined to fulfil the targets set in the respective EU policies (see Table 3, Section 2.2) considering local situations and conditions of each CS (Case Study). Stakeholders may also want to include additional biodiversity targets for certain species that are especially valuable in their region, but are not specifically mentioned in EU policies (Figure 4b). By that, two of the seven principles described in Biggs et al. (2012) would be implemented which are also thought to help increase resilience: (1) broaden participation, i.e. include an adequate number of decision-makers, and (2) maintain diversity, i.e. consider decision-makers with different backgrounds (the remaining five being: manage connectivity, encourage learning, promote polycentric governance, manage slow variables and feedbacks and foster complex, adaptive thinking). Additional opportunities to consider resilience thinking are indicated in Figure 4.

A recent literature review showed how the ecosystem-service perspective is used for setting objectives in freshwater and marine habitat conservation (Boulton et al., 2016).

g. Identifying ecosystem services delivery and demand baselines and objectives

In parallel to defining the baseline biodiversity status and clarifying policy objectives, information on ESS delivery and demand for the respective case study are collected (Figure 4A). The objectives for ESS delivery are defined by stakeholders and/or derived from policies (e.g., from the EU Floods Directive [2007/60/EC]; Figure 4B).

h. Projecting species distributions

Based on the statistical relationship between species occurrences and range-wide environmental variables (see section a above), species probabilistic habitat suitability is projected for each scenario (Figure 4d). In general, scenarios stem from a specific storyline, and build upon potential future socioeconomic patterns, changes, or advances that are translated into potential emissions, CO_2 concentrations, and land-cover change. Thus regarding the models, scenarios are primarily defined as changes in environmental conditions (such as climate change scenarios by the IPCC, 2007), or land cover scenarios (Eitelberg et al., 2015) including the potential fragmentation of the landscape (Leadley et al., 2010). Typically information is available for each spatial unit across the study area, either as a continuous (e.g., temperature) or discrete (land cover type) data[14].

i. Projecting ecosystem services delivery and demand

ESS delivery and demand are modelled for the same scenarios (Figure 4D). To do so, numerous "EBM tools" exist (see https://ebmtoolsdatabase.org/tools) to do so. However, most of them represent ecological, hydrologic, or other biophysical process models that lack

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[14] More information regarding climate-change and land cover scenarios and storylines can be found at http://tntcat.iiasa.ac.at/RcpDb and http://luh.umd.edu/data.shtml#LUH1_Data, respectively.

an explicit focus on ESS. The interdisciplinarity required for the study of ESS is best tackled using integrated modelling tools that are able to represent the wide variety of interactions that happen within SES, such as those based on behaviour, market prices, local versus global economy, etc. In this regard, ARIES [15] is a cyber-infrastructure that integrates multiple modelling paradigms for spatio-temporal modelling and mapping of ESS, supporting artificial intelligence features (semantics and machine learning) for model selection and assemblage to quantify ESS flows from ecosystems to beneficiaries (Villa et al., 2014). To map ESS flow and model its main elements (source, use and sink regions), ARIES uses Service Path Attribution Networks (SPAN = a family of agent-based models) algorithms, which also generate probabilistic spatial outputs and, therefore, enable spatial visualisation of uncertainty (Johnson et al., 2012; see Section 2.6). ARIES currently comprises nine formalised flow types, which serve as a basis for conceptualising flows of other ESS (Villa et al., 2014). As part of ARIES, the initial conditions (i.e., prior probabilities gathered from local experts/stakeholders) can be modelled using Bayesian Belief Networks (BBNs; Barquín et al., 2015). BBNs capture our understanding of the likely cause and effect relationships of multiple influences on a wide range of economic, social, cultural and ecological values (Quinn et al., 2013). The ability of BBNs to concurrently incorporate information from a variety of sources, i.e. empirical data, various types of models, literature and expert opinion, makes them a powerful and flexible method for various applications (Stewart-Koster et al., 2010).

During the last years, ESS mapping tools were moving towards more modelling oriented tools. ESS models are computational representations of the environment that allow biophysical, ecological, and/or socio-economic characteristics to be quantified and explored. When applied to the assessment of ESS, models are important tools that can quantify the relationships underpinnig ESS supply, demand, and flows and, in some cases, produce maps representing these factors. Furthermore, as models can explore scenarios, trade-offs that result from different scenarios can be assessed as well. By means of integrated modelling tools, such as ARIES, ESS mapping can be studied in combination with other ecological and socio-economic interactions that might exert pressures on ecosystems in order to enable EBM approaches. Moreover, in view of ongoing climate change, there is certainly an urgent need to integrate the different elements that compose SES (processes, agents, events, etc.) in order to enhance governance, understand indirect and non-linear causal links, and to be able to predict future scenarios.

In addition, Guerry et al. (2012) showed inVEST as another tool to assess ESS applied to marine ecosystems.

Some examples that illustrate ESS modelling in aquatic ecosystems are those from Lillebø et al. (2016), who applied ESS modelling to case studies in transitional water bodies, Liquete et al. (2016a) in Mediterranean marine and coastal ecosystems, and Arkema et al. (2015), who showed the importance of stakeholder engagement to set management scenarios in marine and coastal ecosystems.

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[15] See http://aries.integratedmodelling.org/

j. Identifying biodiversity and ecosystem services deficits.

Comparing the defined biodiversity targets with the projected species distributions for each scenario as well as the projected ESS with the defined ESS targets will help identify biodiversity and ESS deficits, respectively (Figure 4e and E). With the deficits laid out, actions to improve the situation will then be defined.

k. Identifying action strategies

The combined consideration of the actions needed to overcome impacts on biodiversity (or protect current biodiversity) and the actions needed to provide the targeted ESS delivery, will lead to a set of potential action strategies (Figure 4f). Thereby, resilience has to be taken into account in one of two ways: action strategies are either designed to (i) maintain existing system resilience or to (ii) re-introduce resilience to a system that has previously lost it due to human impacts.

Generally speaking, foresight tools can help planning for the unpredictable (Cook et al., 2014), linking biodiversity and ecosystem functioning (BEF) relationship and resilience of ecosystem and decision-making (Mori et al., 2013). Some illustrations in aquatic ecosystems include those from Baho et al. (2014) showing temporal dynamics, resilience and management options in lakes, and from Mamauag et al. (2013) about targeted adaptation strategies, vulnerability assessment and resilience of fisheries. France (2016) discussed the way to adapt lessons learned from terrestrial ecosystems to marine ecosystems regarding resilience and ESS delivery, in management options (restoration) and governance management.

l. Assessing projected biodiversity and ecosystem services delivery

Species distributions as well as ESS delivery are projected for each of the scenarios x, y and z, considering the expected changes that will result from each individual action strategy (Figure 4g and G). For each scenario, outcomes of the action strategies are assessed according to relevant criteria (i.e., effectiveness, efficiency, equity and fairness, policy implementability) and ranked for biodiversity and ESS separately, depending on the projected benefits they will yield. The highest ranked action strategy for each scenario is marked with a star in Figure 4.

m. Identifying an optimal action strategy

These two assessments and a third assessment of the costs of the individual action strategies are combined, taking into account stakeholders' preferences (Figure 4h). Thereby, action strategies leading to co-benefits or synergies for biodiversity and ESS, i.e. win-win strategies, will be clearly identified as favourable (Figure 4 action strategy 1 in scenario x). An example for a win-win action strategy is the widening of a formerly channelized river course and the subsequent implementation of gravel bars. Such an action strategy has the potential to improve biodiversity (both aquatic and terrestrial) and maintain the river's water purification capacity, i.e. its natural assimilation capacity, among other ESS such as for example an improved potential for recreational activities. In case an action strategy is only beneficial for

certain ESS but has negative impacts for other ESS (especially with provisioning services) or biodiversity, trade-offs need to be incorporated when assessing the optimal action strategy (Figure 4 scenarios y and z; also see Section 2.6). Co-benefits (also called synergies or win-win situations) and trade-offs also occur when targeting different ESS at the same time, whereby trade-offs are three times more likely to occur (Howe et al., 2014). An example for a trade-off among different ESS would be water abstracted for agriculture or human consumption. There are three main indications for a potential trade-off situation: (1) when private gains are connected to social losses of a natural resource, (2) provisioning ESS are involved and (3) if any of the stakeholders is acting exclusively at the local scale. Contrarily, there is no generalisable context for a win-win situation (Howe et al., 2014). The degree to which the assessment of action strategies and the respective trade-offs is guided by stakeholder preferences is case-specific and depends on the defined targets (also see Section 2.6). In any case, the evaluation process considers uncertainties of the projected outcomes of each strategy as well as uncertainties related to the implementation of action strategies (idem). For example, a strategy with moderate benefits for biodiversity and ESS provision may be assessed as being better suited if it leads to a more robust, i.e. less uncertain outcome, as compared to other strategies that potentially have a higher overall benefit but are less certain.

Regarding this assessment, Terrado et al. (2016) suggested a complementary cost-benefit analysis (CBA) to the cost-effectiveness analysis (CEA) in River Basin Management Plans (RBMP) to include the provision of ESS perspective in decision-making. Illustrations in literature showed the way to prioritise cost-effective actions in coastal ecosystems (Giakoumi et al., 2015) and trade-offs cost-effective analysis in river restoration at catchment scale (Hermoso et al., 2012).

n. Optimising the spatial allocation of biodiversity conservation and ecosystem services delivery areas

In case of conflicting effects of the action strategy identified as being the optimal one on biodiversity conservation/restoration and ecosystem service delivery, the allocation of biodiversity conservation/restoration areas and ecosystem service delivery areas may be spatially optimised (Figure 4i). Morán-Ordóñez et al. (2015), for example, used spatial zoning of agriculture and biodiversity conservation areas to increase agricultural development but with a significantly lower impact on biodiversity values and carbon framing than in traditional approaches in Australia's northern savannas. Such a spatially optimised allocation of ESS and biodiversity targets can be done using Marxan with Zones[16] (Watts et al., 2009), which is an extension of the software Marxan (Ball et al., 2009). Marxan solves the so-called minimum-set problem by selecting pre-defined parcels (i.e., spatial units) of land, river or sea from a pool of parcels that together build a conservation network within which pre-defined biodiversity targets (e.g., species or habitats) are protected, while minimising the costs of these parcels and maximising their spatial connectivity. Besides planning conservation

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[16] http://www.uq.edu.au/marxan/index.html?p=1.1.1

networks, Marxan has also been used to prioritise restoration efforts along rivers or within whole catchments (Langhans et al., 2014; Langhans et al., 2016). The major new element in Marxan *with Zones* is allowing any parcel of land, river or sea to be allocated to a specific zone, not just protected or unprotected. To each zone, a specific action, objectives and constraints can be allocated with the flexibility to define the contribution of each zone to achieve targets for pre-specified features (e.g., species or habitats) (Watts et al., 2009). Hermoso et al. (in review) used Marxan with Zones to optimise the allocation of freshwater biodiversity (139 species of freshwater fish, turtles and waterbirds) and compatible ESS (carbon retention and flood prevention by riparian forests and availability of perennial waters) in conservation management zones and areas for accessing provisioning services (groundwater provision for agriculture and recreational fisheries) in trade-offs zones. In doing so, potential trade-offs could be reduced up to 54% (compared to traditional planning), while co-benefits were enhanced up to 26%.

The last step in the full workflow is to monitor the consequences of the implemented, spatially optimised action strategy on biodiversity and ESS delivery. The monitoring will produce valuable information that can feed back into SDMs and ESS models (Figure 4 **a** and **A**). It will also help to refine policy targets and potentially educate and reshape decision-makers' preferences (Figure 4 **b** and **B**), following the concept of adaptive management (e.g., (Kingsford et al. 2011).

2.2   Identifying objectives and deficits

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2.2.1 Introduction

This chapter of the AQUACROSS AF aims at providing insights on how to identify and set local-level, measurable objectives for policy action. Accordingly, this chapter covers how to structure objectives at different levels, from global and EU policy to regional and local scales as well as from abstract, general goals to specific and measurable targets tailored at the local level (see Box 5). Examples from the AQUACROSS Case Studies are provided in text boxes and tables to illustrate the concepts and steps provided in this chapter (see, for instance, Table 2).

As in Section 1.3.5, the definition of objectives builds upon the baseline analysis (see above), where the main challenge and the policy context have been set along policy priorities for the local level. The operational definition of objectives to assess progress at the local level would definitely benefit from the analysis of social drivers of ecosystem change, the resulting pressures and the assessment of the current and baseline status of the relevant ecosystems (Section 2.4), as well as from the analysis of how all this links to biodiversity and ESS (Section 2.5).

The AQUACROSS concept and the previous section stress upon the fact that both levels of objectives (global and local) refer to desired, intended or target conditions of the ecological system, not of the entire SES. At local level, though, objectives should be designed in order to restore the sustainability of the whole SES. Nevertheless, this all-embracing target has a necessary precondition, a key one: reaching a sustainable status of the ecological system. Nowadays, it is common practice that the goals of EU environmental policy (see Table 3) and, therefore, the goals of the relevant Strategies of Directives are stated in terms of conservation, preservation, protection, enhancement of biodiversity, habitats, water bodies, etc. Assessment does refer then to both the ecological and the social systems as well as to complex links among them; primary objectives however only seem to address the ecological system.

Box 5: Example opportunities of global and local action to conserve the future state of marine biodiversity and fisheries

Internationally, leaders must take action to deal with global threats, i.e. climate change impacts, and work on global binding solutions. Opportunity for these actions include:

- Coordinate international efforts to reduce greenhouse gas emissions, and increase the adaptive capacity of developing countries to face climate change impacts on fisheries;
- Stop illegal, unreported and unregulated fishing and ban the use of bottom destroying fishing gear;
- Augment progress in the integration of fishery-depended datasets and research survey datasets so that they are made interpretable and can be pooled for large-scale analyses. This is important, because human threats to biodiversity, including from commercial fisheries, occur across large spatial and temporal scales. Therefore, biodiversity and ecosystem monitoring, forecasting and risk assessments, such as improved understanding of tipping point thresholds, require data to be organised in a global, integrated infrastructure, such as provided by the Global Biodiversity Information Facility and Ocean Biogeographic Information System.

Local and regional governments must take action to stop illegal, unreported and unregulated fishing, in addition to removing harmful fishing subsidies. Opportunity for these actions include:

- Implement comprehensive and integrated ecosystem-based approaches to manage human activities (e.g. ,aquaculture, fisheries, coastal development) in coasts and oceans, and to manage disaster risk reduction and climate change adaptation;
- Reduce fishing capacity and rebuild over-exploited ecosystems; this could be achieved partly by eliminating subsidies to the fishing industry that promote overfishing and excessive capacity;
- Adopt environmentally-friendly and fuel efficient fishing and aquaculture practices and integrate 'climate-proof' aquaculture with other sectors;
- Strengthen knowledge of aquatic ecosystem dynamics and biogeochemical cycles, particularly at local and regional levels;
- Strengthen the adaptive capacity of local populations to climate change impacts by conducting local climate change assessments of vulnerability and risk and through an investment in raising people's awareness, namely in schools and among stakeholders.

Source: Leadley et al. (2010)

Part of the pervasive confusion in EU policies between policy objectives and assessment criteria stems from the above-mentioned pattern of primary objectives referred only to the ecological system, not the social-ecological one. Policy objectives are the primary ends of environmental policy, while assessment criteria are used to judge the system as well as the alternative means that might be used to reach those goals. Within AQUACROSS any policy objective is defined in terms of a desired or target condition of the involved aquatic ecosystem, including its biodiversity. Hence, the analysis of any other ambition related to the social system (ensuring financial responses, avoiding a high degree of social contest, addressing distributional impacts, etc.), should be considered within the criteria to assess the alternative ways to reach the primary environmental targets as well as the institutional potential to meet what is required for sustainability (see Section 2.3).

Table 2: Environmental Challenges of the AQUACROSS Case Studies

AQUACROSS Case Study	Examples of Some Environmental Challenges Found in Case Studies
Case Study 1: Trade-offs in ecosystem-based fisheries management in the North Sea aimed at achieving Biodiversity Strategy targets	Pressures from fishing (extraction of species)
Case Study 2: Analysis of transboundary water ecosystems and green/blue infrastructures in the Intercontinental Biosphere Reserve of the Mediterranean Andalusia (Spain) – Morocco	Organic pollution (nutrients) and water abstraction
Case Study 3: Danube River Basin – harmonising inland, coastal and marine ecosystem management to achieve aquatic biodiversity targets	Morphological alterations to river and coastal habitats
Case Study 4: Management and impact of Invasive Alien Species (IAS) in Lough Erne in Ireland	Invasive Alien Species
Case Study 5: Improving integrated management of Natura 2000 sites in the Vouga River, from catchment to coast, Portugal	Various sources of micro and macro pollutants, invasive Alien Species, alterations to river and coastal habitats
Case Study 6: Understanding eutrophication processes and restoring good water quality in Lake Ringsjön – Rönne å Catchment in Kattegat, Sweden	Organic pollution (nutrients)
Case Study 7: Biodiversity management for rivers of the Swiss Plateau	Various sources of micro and macro pollutants, habitat alteration
Case Study 8: Ecosystem-based solutions to solve sectoral conflicts on the path to sustainable development in the Azores	Pressures from fishing (extraction of species)

2.2.2 Setting objectives: contributing to EU policy objectives

Overall, the EU Biodiversity Strategy has six targets, but fails to provide clear environmental objectives for the purposes of managing aquatic ecosystems at the local level. For example, the Strategy states that Member States should restore 15% of degraded ecosystems by 2020, but there are no clear objectives on how to do so or what actually constitutes a 'restored' ecosystem. Though these targets set forth overarching objectives that are flexible enough in their wording to allow Member States the freedom to implement them in various ways (i.e., suitable for EU level), they fail to provide measurable objectives for local administrators and managers of these systems (i.e., at the local level). As a further drawback, the Strategy makes reference to other EU Directives and their implementation. These Directives each contain their own goals and objectives (see Table 3 and below). Thus, to support the achievement of the Biodiversity Strategy targets, it is necessary to support the implementation and achievement of other environmental Directives and their respective goals and objectives (i.e., achieving favourable conservation status, status of bird populations, good status for all waters, good environmental status for marine waters) for aquatic ecosystems.

Table 3: Goals and objectives of the main EU environmental directives relevant to aquatic ecosystems

	Habitats Directive	Birds Directive	Water Framework Directive	Marine Strategy Framework Directive
Main protected subjects	Natural habitats and wild flora and fauna of Community interest	All naturally occurring wild birds (including their eggs, nests and habitats)	Inland surface waters, coastal waters and groundwater	Marine waters including coastal waters, seabed and subsoil
Goals	Contribute to biodiversity through conservation of natural habitats and of wild fauna and flora	Conservation of all species of naturally occurring birds in the wild state in the European territory of the Member States	Protection and improvement of inland surface waters, coastal waters and groundwater	Achieve / maintain good environmental status (GES) in the marine environment
Objectives	– Maintain / restore favourable conservation status (FCS) of relevant habitats and species throughout their natural range – Designate Special Areas of Conservation (SACs) for the conservation of relevant species – Management of features of the landscape which are of major importance for relevant species – Regulation of deliberate introduction into the wild of non-native species so as to prejudice relevant habitats and species – Avoid deterioration of relevant habitats and disturbance of relevant species in Natura 2000 sites (Special Areas of Conservation [SACs] and the Bird's Directive Special Protection Areas [SPAs])	– Maintain / adapt the population of wild birds to a certain level (corresponding to ecological, scientific, cultural, economic and recreational requirements) – Designate Special Protection Areas (SPAs) for the conservation of relevant species – Regulate that any introduction of species of bird which do not occur naturally in the wild state does not prejudice the local flora and fauna – Preserve, maintain or re-establish a sufficient diversity and area of habitats for all relevant species of birds	For surface waters: – Prevent deterioration of surface water bodies – Protect, enhance and restore surface water bodies to achieve good surface water status – Protect and enhance artificial and heavily modified surface water bodies to achieve good ecological potential and good surface water chemical status – Reduce pollution from priority substances / phase out emissions, discharges and losses of these substances – Ensure that discharges into surface waters are controlled according to a combined approach For protected areas: – Achieve compliance with standards and objectives under protected area legislation – Establish a register of protected areas lying within RBDs (River Basin Districts) *There are other detailed objectives for groundwater and drinking water	– Ecosystems function fully – Ecosystems are resilient to human-induced environmental change – Species and habitats are protected, biodiversity loss prevented – Ecosystem properties support the ecosystems – Anthropogenic inputs do not cause pollution – Achieve qualitative descriptors used for determining GES: biological diversity, non-indigenous species, commercially exploited fish and shellfish, food webs, eutrophication, sea floor integrity, hydrographical conditions, contaminants, contaminants in fish and seafood, marine litter and energy including underwater noise

Goals and objectives, as stated through environmental agreements, conventions and legislation, must be adopted and translated into national legislation by each ratifying country. Each Member State and relevant local authorities that are charged with implementing them must decide upon whether these goals and objectives are taken at face value or are rather seen as minimum requirements for action. Therefore, in practice, the setting of goals and objectives must be founded on those established at the international and/or EU level but tailored to the local level and the stakeholders involved therein.

Tailoring to the local level

Setting specific policy objectives at local level, that are consistent with EU objectives, can be very challenging. They refer to conservation and biodiversity but must take into account the structure and functioning of an ecosystem and its biological components to address a variety of human needs (i.e., ESS used by socio-economic systems) (Tear et al., 2005). Setting objectives also requires knowledge on the main human pressures on ecosystems and the drivers behind represented by the demand of goods and services provided by nature. Drivers are in general linked to multiple human activities that must be coordinated since managing ecosystems and the services they provide require changing underlying decisions (see Section 2.4).

With respect to environmental management and conservation, it is important to understand the role objectives play in shaping where and how limited conservation or management financial resources are spent (Tear et al., 2005). Here, it is useful to distinguish between goals and objectives. The setting of local objectives refer to the particular condition of the ecosystems at hand that guarantee resilience and sustainability; that is to say, they are the status required to fulfil the criteria to assess the full SES (see Section 2.3 below) and they include its resilience, adaptability and transformability. In practical terms, such local objectives can be defined as, for instance, eradicating invasive species, halting eutrophication, etc. or reducing risk in failing to achieve ecological status of a certain level. Whether these objectives are the appropriate objectives to guarantee the sustainability and resilience of the system is unknown, but as discussed, at a local level, objectives need to be defined to respond to a well-defined environmental challenge and this also depends on those responsible for implementation. The rest of this chapter deals with how to define measurable objectives and gives examples of how to assess against these objectives.

Box 6: Examples of objectives from AQUACROSS case studies 8 (Azores) and 3 (Danube River Basin)

Key pressure: fishing (extraction of fish biomass) – but also additional pressures from ships/ferries, agricultural run-off, sand extraction, and some minimal pressures from tourism/recreation (e.g., anchoring, collision with whales) and maybe in the future deep sea mining (but outside of marine protected areas).

Key changes in relation to state indicators: loss or reduction of fish biomass

Key CS question: Balancing trade-offs between fishing (extraction of fish) with tourism (diving/viewing fish), as the activities are competing for the same resource (fish) in the same spaces.

Key socio-economic challenge: shifting from traditional/historical sector fishing to an emerging sector (tourism).

These are just preliminary at this stage.

Another example from the Danube CS could be the broad objectives for hydromorphological alteration from the International Commission for the Protection of the Danube River (ICPDR):

- Enhance longitudinal continuity for fish migration (constructing fish migration facilities, avoiding new barriers, etc.).
- Restoration, conservation and improvements of river morphology, habitats and their connectivity for specified locations.
- Protection, conservation and restoration of wetlands/floodplains to ensure biodiversity, the good status in the connected river water body, flood protection, pollution reduction and climate adaptation for specified wetlands/floodplains.

- Measures addressing hydrological alterations (hydropeaking, water abstraction for specified locations.

Box 7: Example of objectives from the AQUACROSS case study 7 (Swiss Plateau)

The Swiss Plateau is a densely populated area and Swiss rivers are influenced by various human activities. A key objective for river management on the Swiss Plateau is to improve the ecological state of rivers regarding hydromorphological, chemical and biological conditions.

Natural communities of fishes, macroinvertebrates, macrophytes and algae define a good biological state of rivers. However, there are many other societal needs that have to be considered. These include flood protection, agricultural production, waste and storm water disposal, hydropower production, drinking water production, recreational activities, as well as budget constraints for river management. Current management strategies to improve the ecological state of rivers include an improvement of water quality by reducing diffuse inputs of nutrients and plant protection products from agriculture, upgrading of waste water treatment plants to remove micropollutants, relocation of waste water treatment plants to larger receiving waters, river restoration to improve the morphological state, the protection and extensification of the riparian zone, removal of barriers to improve fish migration, mitigation of hydropeaking, and improvement of sediment transport (Kunz et al., 2016).

Describing objectives

Once the objectives have been defined at a local level, in accordance with EU policy goals, the next step consists in making these objectives operational for the assessment of baseline and policy scenarios. Thus, with the information gathered regarding the political and ecological situation of the local area or region of an aquatic ecosystem, local administrators and stakeholders can jointly develop tailored objectives to address the local-level problem previously identified (i.e., an aquatic ecosystem that is failing to meet the targets of the EU 2020 Biodiversity Strategy due to identified local problem).

Reviewing the respective national transposition of the main Directives may be one useful step to get informed about how EU objectives have been detailed at a national and local level.[17]

The first step consists in specifying the general objective of the Directive by describing the characteristics used to describe targets. For instance, the MSFD describes the ecological status of a marine ecosystem by using 11 descriptors. The WFD describes the ecological status of a water body by referring to a wide array of descriptors grouped into three categories (biological, chemical and hydro-morphological status) and each one of these descriptors can be characterised by a set of indicators that can eventually be measured qualitatively or quantitatively so as to allow comparing the ecological status and characterise the baseline (see Table 4).

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[17] Some countries adapted existing legislation to incorporate the changes required under a particular Directive. Thus, while some Member States may have one piece of legislation covering the Habitats Directive, another Member State may spread out these requirements over 13. These documents and their links can be found online for the each Directive: National Implementation Measures (transpositions to national legislation by each Member State): Habitats Directive; Birds Directive; Water Framework Directive; Marine Strategy Framework Directive.

Table 4: Descriptors for assessment within the main EU Directives for aquatic ecosystems

Habitats Directive	Birds Directive	Water Framework Directive	Marine Strategy Framework Directive
Natural habitat types: • Range, • Areas covered, • Specific structure and functions, • Future prospects Species (non-bird): • Range • Population • Habitat for the species • Future prospects	No detailed definition – but similar logic is used as for species under the Habitats Directive.	Detailed in Annex V: • Biological: aquatic flora, macroinvertebrates, fish, etc. • Physico-chemical: nutrients, oxygenation, acidification, salinity, etc. • Hydromorphological: hydrological conditions, continuity, bed substrate, etc. • Priority substances and chemicals relevant for groundwater	11 descriptors in Annex I plus details in Annex III and GES Decision criteria: 1. Biodiversity 2. Non-indigenous species 3. Commercial fish and shellfish 4. Food webs 5. Eutrophication 6. Sea-floor integrity 7. Hydrographical conditions 8. Contaminants 9. Contaminants in seafood 10. Marine litter 11. Energy incl. underwater noise

Source: Rouillard et al., 2016

Figure 5: Example of deficits between baseline and target status of Habitats Directive descriptors
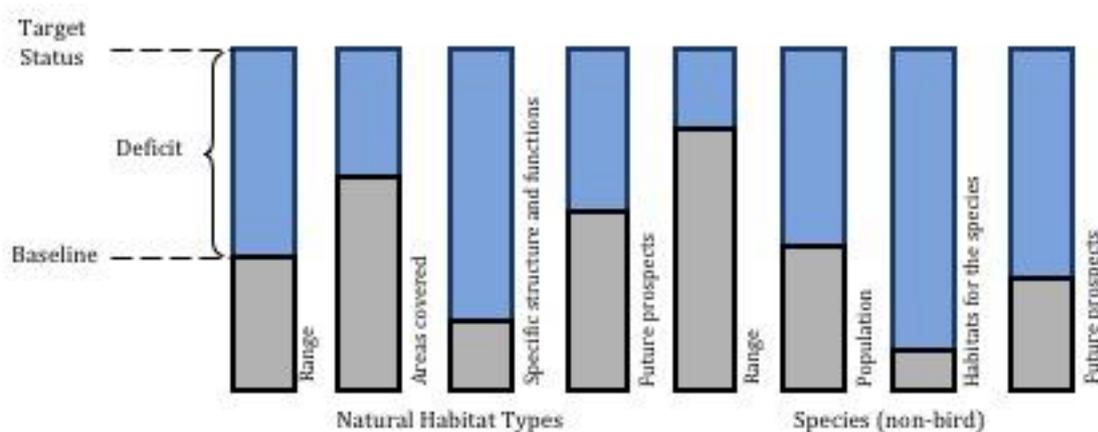
Source: Own illustration.

Legend: The grey teal areas of the bars above represent the current state of a descriptor for the Habitats Directive. As such, these can be the existing range of natural habitat types, the amount of areas covered for these natural habitat types, etc. The blue section of the bars represents the desired state of these descriptors, corresponding to a policy's targeted status for that descriptor, which varies between each descriptor. The distance between the grey teal and top of the blue bar represents the deficit or gap between current state and the desired state (or target status) of each descriptor.

The development of descriptors and indicators allows the specification of the objectives and provides information on the gap between baseline and target status of the implied ecosystems. These distances, sometimes called deficits, are the gaps that must be bridged in order to fulfil the desired objectives. As such, descriptors and indicators can provide a starting point to help local-level managers focus on key aspects of ecosystems under their jurisdiction and develop targeted and measurable objectives to reach a desired status (see Figure 5).

Assessing baselines and setting deficits

Describing policy objectives following the methods developed for EU policy objectives allows the mobilisation of huge amounts of data and the production of research results and implementation at EU, national and local levels. For instance, many reports and assessments are available to determine the current status of the implementation of the EU 2020 Biodiversity Strategy, the Nature Directives (Bird and Habitats Directives), WFD and MSFD within each Member State. Member State assessments and reports for the different Directives can help guide the identification of relevant descriptors and the best sources of information within a region or area. This not only connects the local level to the national level, but also provides an opportunity to integrate higher-level national objectives into local-level environmental decision-making processes.

Sources for EU assessments and Member State reporting on the main Directives can be found online, the links for some of which are listed in Table 5, below. In addition to the policy reviews and assessment of current status of implementation, it is highly important to determine the ecological status of the aquatic ecosystem to be managed. Most of the assessments within the links above should make reference to the ecological state and status of the area in question. Information should be gathered regarding: the current status of the water bodies (chemical status, ecological status, etc.)/initial assessments of ecological state for marine areas/levels of biodiversity and number of protected area sites etc. within the administrative boundaries of a site.

Table 5: Links to major assessments and national reports for the main EU policies relevant to aquatic ecosystem management

Policy	Sources
Biodiversity Strategy	• Mid-term review of the EU's Biodiversity Strategy: European Parliament resolution of 2 February 2016 on the mid-term review of the EU's Biodiversity Strategy (2015/2137(INI)) • Report from the Commission to the European Parliament and the Council: The Mid-Term Review of the EU Biodiversity Strategy to 2020 • Mid-term review of the EU biodiversity strategy to 2020 EU assessment of progress towards the targets and actions
Natura Directives (Birds and Habitats Directives)	• Habitats Directive reporting (information page and links) • The State of Nature in the European Union: Report on the status of and trends for habitat types and species covered by the Birds and Habitats Directives for the 2007-2012 period as required under Article 17 of the Habitats Directive and Article 12 of the Birds Directive • European Environment Agency's "State of Nature in the EU" Technical Report No 2/2015 • Web tool on biogeographical assessments of conservation status of species and habitats under Article 17 of the Habitats Directive • Member State National Summaries for Article 17 of the Habitats Directive (2007-2012) • Birds Directive reporting (information page and links) • Reporting under Article 12 of the Birds Directive (period 2008-2012): *Member State Deliveries*
WFD	• WFD Implementation reports (information page and links) • Links to the official WFD implementation web sites of the EU Member states • River basin Management Plans for the WFD and the Floods Directive
MSFD	• Reporting for the Marine Strategy Framework Directive (information page) • The first phase of implementation of the Marine Strategy Framework Directive (2008/56/EC): The European Commission's assessment and guidance • JRC In-Depth Assessment of the EU Member States' Submissions for the MSFD under articles 8, 9 and 10 • Report from the Commission to the European Parliament and the Council on the progress in establishing marine protected areas (as required by Article 21 of the Marine Strategy Framework Directive 2008/56/EC

Box 8: Example of the elaborated definition of a GES descriptor for the MSFD

Descriptor 5: Eutrophication

GES definition used: GES with regard to eutrophication has been achieved when the biological community remains well-balanced and retains all necessary functions in the absence of undesirable disturbance associated with eutrophication (e.g. excessive harmful algal blooms, low dissolved oxygen, declines in seagrasses, kills of benthic organisms and/or fish) and/or where there are no nutrient-related impacts on sustainable use of ecosystem goods and services.

Risk categories for Eutrophication

High	Undesirable disturbance* caused by eutrophication is widespread (even or patchy) and frequent in the region (> once a year)
Moderate	Undesirable disturbance* caused by eutrophication is widespread but rare in the region (< once a year)
	And/or
	Undesirable disturbance* caused by eutrophication only occurs at a site or local scale in the region, but it occurs at least once a year
Low	Undesirable disturbance* caused by eutrophication does not occur in the region, or where it does occur it only occurs rarely (<once a year) and on a very local scale (site or local patchy)

*Undesirable disturbance includes one or more of the following: harmful algal blooms, low dissolved oxygen, associated declines in perennial seaweeds or seagrasses, kills of benthos and fish, and dominance by

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Source: Own illustration. Developed by ODEMM, followed by risk categories that relate to this (taken from Robinson et al., 2014). *For definitions and risk criteria used for each of the GES descriptors see Breen et al. (2012).

For convenience, specific objectives are often tied to the descriptors of the various Directives (see Table 5), this allows taking advantage of the increasing information and knowledge already developed to support policy-making, administrators and managers to assess baselines and progress towards reaching EU policy objectives.

Comparison between objectives and baseline based on the descriptors of the environmental objectives then allow identifying the gap (deficit) that must be bridged and provides an operational definition of the objectives (e.g., Robinson et al., 2014). This can then be used to assess the effectiveness and facilitate the choice of management measures (e.g., Piet et al., 2015), which will be suited to achieving policy objectives and then to reducing deficits.

For instance, Bouleau and Pont (2015) discussed about policy gaps and deficits related to spatial scale regarding WFD objectives and its operational implementation and assessment within MS, and Stelzenmüller et al. (2013) about policy goals integration in marine ecosystems.

In addition to these challenges, there also are policy failures in recognizing the importance of connectivity among land, inland waters, and seas (e.g., Ormerod and Ray, 2016).

Table 6: Assessing baselines for setting policy objectives: risk of departure from GES for MSFD descriptors for the four European regional seas using the ODEMM GES Risk Assessment

GES Descriptor (and characteristics)	Northeast Atlantic	Mediterranean Sea	Baltic Sea	Black Sea
Biodiversity-Phyto-zooplankton	Low-Moderate	Moderate	Moderate	Moderate
Biodiversity-Fish	Moderate	Moderate	Moderate	Moderate
Biodiversity-Marine mammals and reptiles	Low-Moderate	High	Moderate	Moderate-High
Biodiversity-Seabirds	Moderate	Moderate	Moderate	High
Biodiversity-Predominant habitat types	Moderate	Moderate	High	Moderate-High
Non-indigenous species	High	High	High	High
Commercial fish and shellfish	High	High	High	High
Food webs	High	High	High	High
Eutrophication	Moderate	Moderate	High	Moderate
Sea floor integrity	High	High	High	High
Contaminants	Moderate	Moderate	Moderate-High	Moderate-High
Contaminants in fish and shellfish	Low-Moderate	Low	Moderate	Moderate
Marine litter	High	High	High	High
Underwater noise	High	High	Moderate-High	High

Source: Breen et al. (2012)

Figure 6: Example of a hierarchy of environmental policy objectives
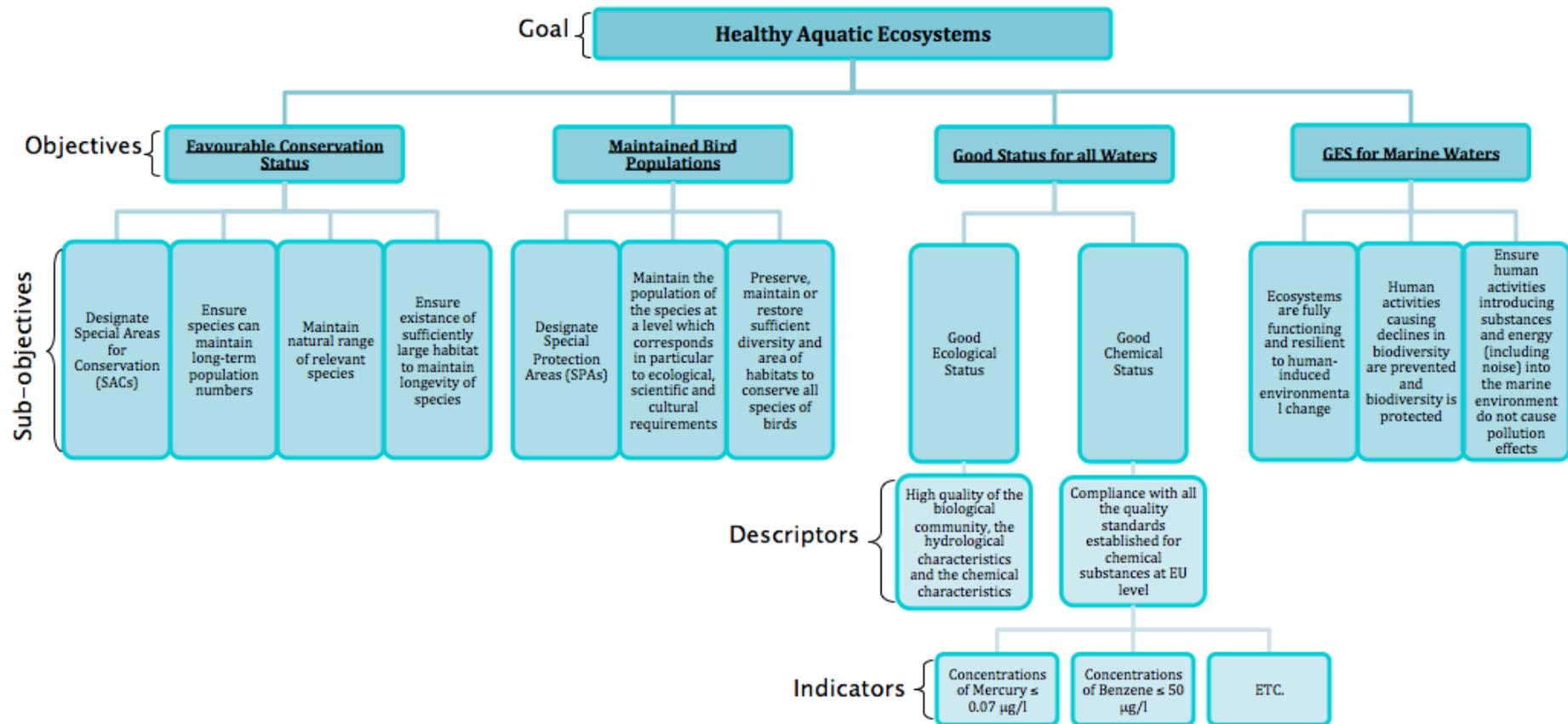
Source: Own illustration

2.3   Assessment criteria: a key element to assessing baselines and policy options

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2.3.1 Assessing baselines and policy scenarios

Taking into account the whole AQUACROSS Architecture (above) and Section 1.3.6, this chapter will provide the basics for making resilience thinking operational to assess current and prospective baselines as well as alternative policy scenarios. It will include specific criteria such as adaptability, transformability and others that may be relevant to assess the sustainability of ecological and social systems within the AQUACROSS framework.

To make a holistic approach operational the first basic requirement consists in being able to assess the whole social–ecological system, in general, and its foreseeable trajectories, in particular, under broad sustainability criteria; that is to say, according to the AQUACROSS innovative concept (Gómez et al., 2016), based upon resilience thinking.

Making resilience thinking operational to assess the sustainability of both baseline and policy scenarios entails judging the social and ecological systems as well as their mutual interactions according to the three attributes or assessment criteria that determine the future trajectories of the social–ecological system: its **resilience** *per se*, its **adaptability**, and its **transformabilit**y.

<u>Resilience</u> refers to the capacity of a system to deal with disturbance and continue to develop (Folke et al., 2010). Resilience is therefore defined as a measure of the amount of perturbation a linked social–ecological system (SES) can withstand and still maintain the same structure and functions (Holling et al., 2002; Walker et al., 2004). Following Hill et al. (2014), there has been a growing body of work identifying traits of adaptive governance and management that enable a system to manage and cope with increased uncertainty in dynamic systems and changing social–ecological baseline conditions: including *flexibility* in social systems and institutions to deal with change; *subsidiary* and *connectivity* (openness of institutions providing for extensive participation, effective multi–level governance); *iterativity* (social structures that promote learning and adaptability without limiting options for future development).

In the terms of the AQUACROSS Architecture this refers to the capacity of the social–ecological systems to co–produce the ecosystems services and abiotic outputs that would be demanded by society in the long term.

<u>Adaptability,</u> one basic attribute of resilience, refers to the capacity of actors in the system to manage change so as to maintain the system within sustainability boundaries. Adaptability

reflects the capacity of a SES to learn, combine experience and knowledge, adjust its responses to changing external drivers and internal processes, and continue developing within the current trajectory (Berkes et al., 2000), maintaining or enhancing inclusive wealth (Chapin et al., 2009).

As a result, adaptability contributes to mitigate uncertainty by corrective action. This entails for instance the capacity to adapt pressures over ecosystems to conditions that cannot be anticipated with certainty. This is, for instance, the case of the relative abundance of commercial fish species at any moment in time, changes in rainfall and runoff across different spatial and temporal scales, exposure to water-borne risks such as floods, droughts, landslides, etc. and progressive detrimental trends in water pollution, scarcity, fish exhaustion, etc. Adaptability also includes refers to changing policies or management practices when new knowledge becomes available, and putting processes into place that allow for continuous evaluation and learning through experience and sharing of different understandings.

One critical objective of policy actions within AQUACROSS consists in enhancing the robustness of the system, that is to say its capacity to absorb shocks and adapt to circumstances that are not completely predictable in advance.

In the AQUACROSS project adaptability becomes the central assessment criterion to assess the capacity of the social system to respond to new circumstances. This implies for instance a critical analysis of technological development, a process increasingly driven by scarcity of critical resources such as water and energy and by the increasingly impaired capacity of ecosystems to deliver services such as fish biomass and water, or nutrient absortion. Recent technological progress in aquaculture, water efficient irrigation, and microfiltration for wastewater treatment or desalination are just some of the most relevant adaptive responses provided by technology.

Sometimes, though, adaptation is not enough to ensure the sustained and sustainable production of desired ESS, for instance in situations where ecosystems have gone through certain thresholds or social processes are too entrenched to be easily adapted. In those cases a disruptive change may be necessary that leads to fundamentally different ways of using and managing ESS (and possibly different ESS in the first place).

Similarly adaptability can be linked to the capacity of economic agents to take the best available technologies that, due to their potential in reducing pressures over ecosystems, may support the compatibility of the eventual expansion of the economy with the preservation and improvement of ecosystems. According to this, one important barrier to adaptability has to do with technological lock-ins, that is to say mechanisms that maintain water-related activities locked into traditional practices and technologies due to transaction costs, risk-avertive behaviour or evidence of wrong incentives. Technology and technological systems in water management follow specific paths that persist even in face of competition from more effective and efficient alternatives (Rip and Kemp, 1998; Gandhi and Crase, 2012; Garrick et al., 2013; Jain and Gandhi, 2016).

Besides barriers to innovation and the limited capacity to uptake innovations once they are available, adaptability critically depends on the capacity of institutions to grow to the challenge of screening, designing and implementing policy responses that are better adapted to new circumstances so as to overcome traditional responses. This is particularly relevant for the application of innovative management approaches, such as EBM. Prevailing institutional set-ups are better fitted for the design and implementation of traditional management alternatives.

For instance, fish policy is much better adapted for managing commercial fisheries through regulating fishing effort than to manage the complex marine ecosystems on which the supply of fish and many other ESS depend. Similarly, freshwater institutions are better designed to manage individual services such as the provision of water for households and economic activities such as agriculture, manufacturing or tourism than to actually govern freshwater ecosystems that provide these services along with many others, such as runoff regulation, flood control, biodiversity support, etc. that may well be overlooked (Colding et al., 2006; Wamsler et al., 2016; Al-Saidi and Elagib 2017).

Likewise, current management of aquatic ecosystems is fragmented into different institutional silos specialised in particular economic sectors (water utilities in urban areas, irrigation districts, fish management authorities, land planning offices, storm management agencies, nature preservation authorities, etc. at different government levels with little (if any) communication amongst them (see for instance Laborde et al., 2016 for inland fisheries and Dieperink et al., 2016 for flood management). There is increasing evidence that this fragmentation reduces adaptability by the simple fact that it makes the governance system in place unable to take advantage of the synergies of EBM approaches. EBM, by focusing on the restoration and protection of ecosystems, yields benefits over a wide range of policy domains (as in the case of river restoration measures that improve the river ecology, reduce flood risk, contribute to habitat preservation objectives, reduce wastewater treatment costs, etc.).

In fact, the lack of institutional coordination may be one of the leading causes of aquatic ecosystems degradation, as different policy areas generate incompatible demands over the services provided by the same ecosystem. Such is the case of concurrent advances of agriculture, land settlements, energy development, manufacturing, etc. that ignore each other's demands in their baseline scenarios and result in aggregated pressures that cannot be met by the water ecosystem at all.

Adaptive management, like any iterative learning and decision process, addresses prior information, decision-making, and observed consequences, which are not final events but rather new sources of information that may lead to changes in management practices (Ascough et al., 2008). For instance, Summers et al. (2015) described an Adaptive River Management framework, related also to effectiveness and implementability criteria.

Transformability refers to the capacity to create a new system when ecological, economic, or social structures make the current system untenable (Folke et al., 2010). This is, for example, the case of water stressed regions that, once the opportunities of water development through

the construction of infrastructures have been exhausted, need to re-examine the way they manage water with the use of demand management alternatives able to meet all demands from all areas of the economy within available resources. This is also the case of the fishing sector that may need to move the main focus away from promoting investments in fishing gears and towards conservation of fishing stocks and reducing fishing efforts once sustainable yields have been exceeded.

Transformability addresses active steps that might be adopted to change the system to a different, potentially more desirable, state. It includes actions to identify potential future options and pathways to get there, the capacity to learn from crises and to navigate thresholds for transformations (Chapin et al., 2009).

Developing the capacity to coordinate actions and overcoming the above-mentioned problems of technological and institutional lock-ins are essential changes to transform the capacity of the social system to new, more demanding, ecological challenges.

2.3.2 Assessing the environmental and welfare outcomes of baselines and policy scenarios

This section will define a set of differentiated criteria to assess policy outcomes for both EBM and traditional responses and will provide basic guidance to make them operational within AQUACROSS. The section is organised according to particular criteria.

Effectiveness: reaching the environmental target

The first obvious way to assess the performance of a social system, both in baseline and policy scenarios, consists in determining, and eventually measuring, how close or far they get to commonly agreed environmental outcomes. Effectiveness, as a policy evaluation criterion, refers to indicators of the public and individual decisions to keep up to the promise of reaching good status or any other precisely defined objectives across water-related policy domains (see Section 2.2 on policy targets).

The accuracy of the analysis of effectiveness of individual measures, integrated approaches or simply in the baseline scenario, depends on how precise policy targets are. For instance, when the desired status of ecosystems is defined through a list of indicators (e.g., reference conditions of good status of water bodies under the Water Framework Directive) it becomes possible to measure policy targets in terms of the gap that must be bridged between baseline and reference conditions (such as water quality, quantity, hydromorphology or any other set of attributes) and alternative courses of action (see Section 2.5 on indicators of the status of conservation of ecosystems and biodiversity).

Methods to assess and compare the effectiveness in reaching policy targets need to be adapted to account for the specificities of EBM when compared against more traditional approaches. Conventional measures are more specialised than innovative EBM approaches. For instance, freshwater quality problems can be managed by collecting effluents and

diverting them to wastewater plants; water scarcity can be managed by reducing leakage in the water distribution networks or by installing more water-efficient devices; controlling waterlogging as a result of storms can be managed by building storm tanks to buffer excess water. The effectiveness analysis of these specialised measures is relatively simple as each individual alternative is better fitted for just one and different end.

Contrariwise, EBM alternatives are not specialised: when properly designed and implemented, they yield benefits over a range of policy objectives. For instance:

▸ Extensive land application systems, as a means to treat wastewater, do improve water quality at the same time as saving energy, capturing carbon, holding soil erosion, supporting the production of biomass, and eventually the restoration of native landscapes (Ortuño et al., 2011; Villar et al., 2011; Sanz et al., 2014).

▸ Green infrastructures in cities are appropriate to control excess storm water whilie simultaneously recharging groundwater resources, restoring urban parks, controling temperature, and saving energy, amongst other benefits (see, for instance, Jaffe, 2011; Longo et al., 2012; Nurmi et al., 2016).

▸ Sustainable soil conservation practices, instead of heavily engineered farming on artificial soils, increase water retention in the soil, thus reducing exposure to droughts and floods and, while allowing for natural soil formation, reducing production costs, increasing yields, maximising soil organic carbon deposits, recharging groundwater, improving water quality, and supporting biodiversity (see for instance an overview of 'conservation agriculture' practices in Palm et al., 2014).

▸ Similarly the interconnection across water realms, at the core of AQUACROSS, implies that the effectiveness of measures or strategies span beyond water bodies that are directly managed. River restoration offers wide evidence on how measures taken in the headwaters have beneficial impacts downstream and, for instance, the improvement in the amount and the quality of rivers may have positive impacts over biodiversity in the river mouth, avoid the formation of death zones, control the proliferation of invasive species, reduce erosion, and improve sediment flows for beaches and ecological niches with positive impacts on human welfare and biodiversity (e.g., meander reconnection and hydromorphological measures and effects on macroinvertebrate community composition in Lorenz et al., 2016a).

In addition, in contexts of multiple stressors, decisions have to be balanced effectively. That requires the identification and valuation of trade-offs between the status of water bodies and the effort required to achieve it - financial, social, or technical difficulties of acting on a single specific pressure - (Pistocchi et al., 2016).

Efficiency: making the most for human welfare

Efficiency is an overarching criterion referring to the capacity of individuals, institutions and social systems overall to make the most out of available resources, including human and natural capital, technologies, infrastructures, etc., in order to improve human welfare. Strictly speaking efficiency is a normative criterion to judge allocations of resources across time and space in terms of their contribution to human wellbeing.

The definition of human wellbeing is far more ambitious that the notions used by individuals when making decisions in markets. For example, while businessmen care for the profits they get, households thrive to make the most of their incomes and farmers to get the highest and safer levels of income at the end of the harvesting period, these are but financial, short-sighted targets that ignore many aspects of human welfare.

Some of those dimensions of human welfare are negative externalities stemming from the degradation of the environment, or of the non-rival and non-excludable goods and services provided by ecosystems such as water regulation, health control, and cultural services, just to mention a few, while others are positive externalities or benefits. Currently, neither of these is reflected in market prices or financial accounts. To counter this, rather than on market profits accrued to individuals, economic analysis must focus on collective benefits and human wellbeing within the social system in order to make the ambitious concept of efficiency truly operational.

The best way to define an efficient market or institution is through its capacity to take advantage of opportunities to make someone better off without making anyone else worse off. From a long-term perspective the notion of efficiency can be closely connected to that of sustainability. In fact, efficiency means that each generation should do its best out of available opportunities to improve its wellbeing as far as this does not result in diminishing the options of future generations.

The first obvious application of the notion of economic efficiency in the face of a policy challenge, such as those identified in Section 2.2, consists in using it to discern whether there are alternative courses of action that might result in higher benefits to some and that do not imply making others worse off. All these opportunities do exist in many cases where, for example, water users could benefit from more water-efficient technologies. Reducing the waste of valuable resources is an obvious way to get more goods and services in the economy without increasing pressures over ecosystems.

Nevertheless this criterion that no one can negatively affected as a result of a change seems too stringent. There are winners as well as losers from most environmental policy strategies, whether traditional and innovative, and all of them, despite the size of their benefits, also face some opportunity costs. This is why weaker but more operational definitions of sustainability rely on cost-benefit analysis and state an alternative situation is superior if the benefits of moving away from baseline trends are higher than opportunity costs.

Box 9: Welfare enhancing opportunities of coping with overfishing at local and regional scales

- Implementing comprehensive and integrated ecosystem-based approaches to manage human activities (e.g., aquaculture, fisheries, coastal development) in coasts and oceans, and to manage disaster risk reduction and climate change adaptation;
- Reducing fishing capacity and rebuilding over-exploited ecosystems; this could be partly achieved by phasing out subsidies to the fishing industry that promote overfishing and excessive capacity;
- Adopting environmentally-friendly and fuel-efficient fishing and aquaculture practices and integrate 'climate-proof' aquaculture with other sectors;
- Strengthening our knowledge of aquatic ecosystem dynamics and biogeochemical cycles, particularly at local and regional levels;
- Enhancing the adaptive capacity of local populations to climate change impacts by conducting local climate change assessments of vulnerability and risk and through an investment in raising people's awareness, namely in schools and among stakeholders.

Source: Leadley et al., 2010.

Some examples of welfare enhancing opportunities that can be identified by applying the above mentioned efficiency criteria are the following:

▶ Distant marine protected areas may entail opportunity costs for local populations, they stilll result in increased benefits worldwide not only for the intangible values of biodiversity but because they play a key role in the biological productivity that results in better fishing opportunities in many other places.

▶ Soil conservation practices in agriculture result in short-term losses to farmers but may also yield benefits to others that spread all over the river basin as a result of improved freshwater quality, runoff attenuation, recharged aquifers, reduced erosion rates, etc.

These efficiency gains are precisely one distinctive feature of EBM, when compared to more traditional approaches. As in the two cases mentioned above, the negative outcomes may be borne by locals (fishermen and farmers) but the better state of conservation of the ecosystem (let us say at the level of the new marine protected area or the farming area) will provide benefits that spread over multiple beneficiaries at regional or even global scales. Comparing these costs and benefits is often challenging because while costs, such as farmers' and fishermen's losses are relatively easy to measure (in terms of foregone income), their benefits are intangible, cannot be measured directly through market prices, and are most of the time uncertain. When assessing EBM approaches under efficiency criteria one should be aware of the difficulties in comparing costs (that can be monetised and are relatively certain) with benefits (that cannot be monetised and are more uncertain).

Nevertheless, given the uncertainties about future conditions, bounded information and the snags of valuing non-market benefits, developing a full-fledged cost-benefit analysis to compare baseline and policy scenarios is barely feasible. This is why an even weaker concept of efficiency can be used as a first approach to economic efficiency: this practical option is the so-called cost-effectiveness criterion.

The rationale of the cost-effectiveness criterion is based upon the impossibility of setting policy objectives based upon optimisation models. Although it may have been the ambition of many economic and ecological normative models, outside of abstract models, there is no way to define things such as an optimal status of aquatic ecosystems (as the one that delivers the most valuable combination of ESS in a sustainable way). Accepting this implies that rather than the optimal outcome of an omniscient and perfect foresight modelling efforts, environmental objectives are social choices that, at least a priori, should not be assessed using efficiency criteria. Hence, the analysis should shift from setting the best possible ends to choosing the best possible means to reach previously agreed policy goals. In other words, adopting least-cost solutions, rather than best-value ones. Indeed, in this context, the best possible strategy is the one that allows reaching the target at the least possible opportunity cost. Well-defined objectives along with good indicators of policy targets are functional to the design of cost-effectiveness indicators that may allow comparing different courses of action and support the choice of the most cost-effective set of measures.

EBM approaches, in many policy relevant contexts, can be proved as the most cost-effective way to meet well-defined policy targets:

▸ Many research projects in the EU as well as practical actions have demonstrated that giving "room to rivers" by restoring floodplains is a more cost-effective way to reduce flood risk than for instance dikes, flood defences, and other heavily engineered alternatives. In addition to being more cost-effective for controlling runoff, restoring floodplains come along with significant co-benefits that would be foregone should other non-EBM alternatives were taken (see DG.Env/D.1-Ares, 2011). These benefits include improved water quality, recharged aquifers, wildlife habitat, recreation, sustainable agriculture, reduced insurance and recovery costs, forestry benefits and carbon sequestration (see 'Nature Conservancy').

▸ Sustainable urban drainage systems (SUDS), that make use of functions traditionally performed by nature, such as water infiltration and runoff regulation, have resulted in savings of billions of dollars as compared to other grey storm management infrastructures (see Sample et al., 2003).

▸ Decisions between the development of a wastewater treatment plant and nature-based solutions such as natural water retention measures (NWRM), should account for investment and operational costs but also for business and social impacts (e.g., the reconnection of a floodplain may entail relocation of economic activities; Pistocchi et al., 2016).

▸ Factoring the value of ESS into decision-making helps identify and negotiate trade-offs between different management options (e.g., short-run agricultural production versus water quantity and quality regulation), and to develop policies to align private incentives with societal objectives, Engel and Schaefer, 2013).

▸ Voluntary agreements to preserve upwaters, favoured by side payments, may result in important savings from an improved supply of more reliable and better quality water

downstream (as compared to better water treatment, desalination plants, and other infrastructures in addition to landscape, afforestation and other environmental benefits).

▸ Marine management measures have been progressing in conjunction with economics – e.g. whaling quotas – towards the enhancement of marine ecosystems and the welfare of coastal human populations. Major challenges to assess the efficiency of EBM are still in place though (Sumaila and Stergiou, 2015).

One of the important hypotheses of a project such as AQUACROSS is that there are relevant welfare enhancing opportunities in improving ecosystems and biodiversity. In other words, as shown in the examples mentioned in the last two sections, improving the status of ecosystems and their functioning may be, in many cases, a more efficient alternative that prolonging current practice, and these advantages can be convincingly shown by assessing the efficiency of EBM alternatives.

Equity and fairness: sharing the benefits

To be fair is as critical as being effective and efficient. This requires definitions and guidance to assess alternative policy pathways in terms of their foreseeable impacts over income distribution as well as for the identification of beneficiaries (linking effectiveness with equity criteria) and benefits (linking efficiency with equity) of both EBM and traditional approaches. Links with other criteria imply trade-offs with efficiency and synergies: sharing the benefits increases feasibility and strengthens enabling conditions for cooperation.

Equity criteria contribute to the assessment by managing complexity and uncertainty, while recognising the diversity of perspectives and knowledge of those affected (Richter et al., 2015).

2.3.3   Assessing governance: growing to the challenge of making EBM happen

According to the AQUACROSS innovative concept (D3.1), EBM focus aims at enhancing, restoring and/or protecting the ability of ecosystems to contribute to sustainability through the continuous provision of a valuable set of ESS, when facing either gradual changes or sudden and unexpected perturbations. It includes strategies to maintain and restore natural ecosystems, protect vital ESS and reduce water and land degradation and the management of habitats to ensure reaching biodiversity targets (see Gómez et al., 2016 and the previous Section 2.2 on the policy objectives of AQUACROSS). Though EBM approaches are designed to improve the structure and function of an ecosystem in order to enhance its resilience, their outcomes are assessed against criteria linked to human wellbeing such as sustainability, efficiency, equity, etc. (see previous section).

EBM approaches differ from traditional management approaches that are not rooted in holistic approaches over social-ecological systems.

In order to stress this difference, AQUACROSS will provide examples of policy failures linked to common practice and also evidence on the consequences of ignoring critical linkages as well as the interaction between multiple stressors.

Criteria under this category are connected with the capabilities and the reforms required to speed up the uptake of EBM approaches as well as to reduce transaction costs by adapting the institutional set up and improving screening, implementation and design of measures and packages of measures.

This includes the analysis of the following relevant criteria. Governance frameworks are adapted to current practice and their adaptation to the requirements of making innovative EBM approaches happen is an integral part of the implementation challenges faced by all the AQUACROSS case studies. The governance-enabling factors that are required for the implementability of EBM have been identified in the AQUACROSS concept as the EBM principles (see Long et al., 2015; Gómez et al., 2016). The assessment of these principles at the case study level may provide the basis to assess the ability of the governance framework in place to hasten further implementation of EBM approaches.

EBM principles can be grouped into three governance requirements to make EBM happen (see Table 7 below) at three different levels: policy, science and management.

▸ First, EBM requires rewiring the objectives of collective action and reshaping of the policy-making processes.

▸ Second, the distinctive nature of EBM poses new demands over science and knowledge.

▸ Third, EBM requires radical changes in the way ecosystems and their services are managed

Table 7: The EBM principles organized along the three domains, i.e. Policy, knowledge and Management, considered for EBM

Domain	EBM Principle	Comment
Policy	Sustainability	These principles apply only to the policy domain where sustainability along all three axis, i.e. ecological, economic and social, is a first requirement of any policy objective that should drive EBM
	Develop Long-Term Objectives	
	Decisions reflect societal choice	These principles all reflect that decision-making towards a fundamental collective agreement on the set of policy objectives and their relative importance needs to be inclusive involving all the relevant actors, specifically science
	Stakeholder Involvement	
	Use of Scientific Knowledge	
Science	Use of All Forms of Knowledge	This is where science should interact with other stakeholders in order to include all relevant knowledge
	Consider Ecosystem Connections	These principles only involve the natural sciences
	Ecological Integrity and Biodiversity	
	Account for Dynamic Nature of Ecosystems	
	Recognise Coupled Social-Ecological Systems	Additional principles that require the knowledge base to cover the entire social-ecological system. This is linked to the "Integrated Management" principle in the Management domain
	Interdisciplinary	
	Consider Cumulative Impacts	
	Acknowledge Ecosystem Resilience	This principle requires consideration of an additional aspect of the social-ecological system
	Appropriate Spatial and Temporal Scales	This principle needs to be considered in relation to all the previous principles within the Science domain
	Consider Effects on Adjacent Ecosystems	This principle covers all the fluxes and influences from outside of the boundaries of the natural ecosystem. This is linked to the "Distinct Boundaries" principle in the Management domain.
	Acknowledge Uncertainty	This principle is most relevant for the interaction of science with the Management domain, specifically the principles "Adaptive Management" and "Apply the precautionary approach"
Management	Organizational Change	These generic principles apply to the design of the EBM
	Distinct Boundaries	
	Integrated Management	
	Adaptive Management	These principles are related to the "Acknowledge Uncertainty" principle in the Science domain
	Appropriate Monitoring	
	Apply the Precautionary Approach	
	Consider Economic Context	This principle can only be applied in management if the knowledge base can provide the necessary information, e.g. principle "Recognise Coupled Social-Ecological Systems"
	Use of Incentives	This principle encourages management to go beyond the "command and control" top-down management, economic and/or social incentives and is partly linked to the "Consider Economic Context"
	Explicitly Acknowledge Trade-Offs	Both principles mostly apply to the interaction between management and decision-making
	Commit to Principles of Equity	

Source: Rouillard et al., 2016

EBM requires resetting policy objectives and processes

According to the AQUACROSS Concept (Gómez et al., 2016) the main governance challenge lies in preserving the capacity of social-ecological systems to remain within a certain range of conditions to meet collective and individual development goals, and to ensure the continuous provision of a desired set of ESS upon which we, and our economy, depend. In the face of ongoing changes and their uncertain future consequences and given the inescapable exposure to uncertain shocks, the key to sustainability consists in enhancing the resilience of the whole social-ecological system (SES).

EBM requires rewiring policy objectives to the long-term objective of sustainability. This is equivalent to building the resilience of the whole SES as a means to make human development sustainable (Biggs et al., 2015a). This implies involving social actors so that they are able to take part in cooperative decision-making in order to consider the multiple trade-offs.

Overcoming traditional practices requires mainstreaming new policy objectives linked to the resilience of the whole SES rather than to the use and provision of singular ecosystems services. These new objectives are linked to resilience (see Section 2.3.1). Since adaptability and the transformability of the system have been largely ignored in traditional policy decisions, traditional choices have resulted in reduced diversity and heterogeneity. Market conditions have favoured most profitable crops and species at the expense of less productive ones. Land use practices, driven by policy and market forces, have fostered uniform ecosystems at the expense of valuable environmental services such as water regulation, pollution control, health security, or biodiversity support.

These practices are nevertheless vulnerable to a change in current environmental conditions as they promote the reduction of biodiversity levels interfering with EF and ESS delivery (Altieri, 1999). For instance, the transition to simplified invertebrate fisheries, favoured by fishing practices aimed at maximising the production of targeted species, has triggered severe shifts to ecological states that are undesirable against both ecological and economic criteria. Further, they accelerate biodiversity decline in broader marine areas, threaten food security and leave remaining species exposed to the risk of collapse due to disease, invasion, pollution and climate change (Howarth et al., 2014).

Deepening implementation of EBM so as to enhance sustainability implies facing more complex decisions that require more inclusive and accountable stakeholder involvement as a pre-condition to deal with new and more important trade-offs among policy objectives and vested interests as well as to enhance the social dialogue and the cooperation required to take advantage of new opportunities.

Rather than technical choices, EBM decisions must be based on social priorities defined in the policy-making process. Trade-offs stem from different sources such as the conflicting interests amongst stakeholders, the balance between short and longer term benefits, the need to forgo current rents in exchange of future security, or between the local opportunity costs and regional and global benefits. Restoring or preserving the ability to absorb change,

far from being free, might have sizeable opportunity costs in the short term that should ideally be weighed against long-term benefits of sustainability. For example, soil conservation practices may contribute to resilience by reducing flood and drought risks (through natural water retention), by stabilizing farmers' income and might also have significant co-benefits in terms of water quality and terrestrial and aquatic biodiversity. Nevertheless, they might also reduce crop yields while increasing production costs and exposure to pests (Rodríguez-Entrena et al., 2014).

Adaptability also implies a tension between the benefits of adapting economic and social decisions to current priorities and demands and/or preserving the options to the future in order to maintain sufficient variation to respond to new environmental challenges (Norberg et al., 2001; Levin et al., 2013). For instance, when improving the connectivity and decreasing the intensity and frequency of flooding in urban floodplain restoration, there are trade-offs with drinking water production as the risk of contamination might increase (Sanon et al., 2012). Similarly, building dikes to cope with flood risk would increase short-run resilience to small periodical floods and investment security, but would not be effective at all to tackle large floods making the same people more vulnerable to climate change in the long run (Palmer et al., 2008). All those are complex decisions that can only be the outcome of inclusive, transparent, and accountable policy processes.

Besides adaptability of policies, there are significant challenges in potential gaps between legislation and institutional capacity, and the ability to implement and enforce the law. Subsidiarity, coordination, monitoring and data provision are crucial to policy implementability (Hill et al., 2014). For instance, Ramírez-Monsalve et al. (2016) reviewed the implementability of management measures framed by the Common Fisheries Policy and the MSFD, supported by coordination and an appropriate science-policy-society interface.

EBM is science-based management

Additionally, advancing towards EBM to overcome current practice requires distinctive demands from science and all kinds of knowledge. Regarding science, EBM requires the design and implementation of innovative research strategies able to deal with crucial methodological challenges involved in operationalizing the resilience thinking approach. EBM requires going many steps further than the specific and bounded models that have traditionally informed current practice and pose over science the demand of new kinds of knowledge able to provide a basic understanding of the complex ecological and social links that have either been overlooked or ignored by traditional management decisions.

Instead of partial analyses that focus on flagship species, hotspots, single pressures, specific impacts, etc., EBM requires focusing on biodiversity and ecosystems. Thus the real possibility of overcoming current practice and their outcomes, in terms of degraded resilience and increased ecosystems' vulnerability, critically depends on the availability of new scientific knowledge to inform new decisions based upon the integrity of the ecosystem.

Likewise, the possibility of breaking down the institutional silos on which sectoral and uncoordinated policies are defined and implemented – sometimes looking at conflicting

targets in water, energy, food, land management and other policy domains – requires new sorts of science and knowledge able to make visible the co-benefits linked to the improvement of a given ecosystem's condition. This integrated knowledge is a pre-condition for designing cooperative instruments and taking advantage of new opportunities (i.e., current research on the water, energy, food and climate change nexus; cf. Biggs et al., 2015a and b; Howells et al., 2013 or the recent interest in the contribution of nature-based measures for EU policies on biodiversity, freshwater or the marine environment (cf. EC, 2012).

EBM also requires new scientific knowledge to better deal with the intrinsic uncertainties of social and ecological systems (see Section 2.6.2). Unlike well established models that attach to a basically deterministic perception of future challenges and look for optimal solutions, EBM acknowledges irreducible uncertainties and the importance of building adaptation capacities not only through restoring critical ecosystems but also building social abilities to respond to a range of possible futures as well as to preserve the option to make decisions adapted to what may prevail in the future (e g. Marshall, 2013; Lukasiewicz et al., 2015).

In addition, available models and tools are not currently those that are required to make EBM possible. Most policy models are designed to maximise the provision of some ESS (drinking water, water for irrigation, urban soil, dilution of pollutants, etc.). In contrast, EBM seeks to maximise the value of natural assets; in other words, the aggregated value of all the flows of ESS it could provide in the future. As far as traditional management has gone too far in transforming ecosystems for a single purpose, the emerging strategies find their more relevant opportunities in the benefits attached to restore natural features as for example to reduce flood risk, to contribute to groundwater recharge or soil formation, to improve water quality or to support life and other simultaneous benefits linked to the recovery of ecosystems' structure and functions (EC, 2015).

EBM requires radical changes in management

EBM requires institutional changes, in order to build cooperation to foster collective action, to share the array of ESS obtained across different stakeholders and policy domains and to break institutional silos along with disciplinary borders and short-sighted, short-term, commercial interest. Whilst traditional measures can be (and have been) effectively implemented in a variety of governance setups, EBM can only be the outcome of robust institutions. Gradually improving current decision-making processes is an integral part of building individual and collective capacities and improving governing institutions is an integral part in the transition towards enhancing sustainability. In other words, the effective implementation of EBM requires adapting prevailing institutions and policy-making processes and overcoming significant barriers to be able to meet policy-making challenges such as:

▸ First of all, defining the objectives of EBM. This requires an identification of what set of ESS may be sustainably provided and their relative importance. As these services are asymmetrically valued by different users this implies trade-offs. As a matter of fact, these trade-offs are pervasive and inherent to any resource management decision. What is special about EBM is that this approach gives prominence to this social decision. It thus

favours transparency and a better framework to confront people, businesses and governments with the consequences of their own decisions.

▸ Second, balancing trade-offs implied in finding the best way to meet any environmental objective. As above, the defining components of resilience and the trade-offs amongst modularity, connectivity, heterogeneity and redundancy, should be considered in the decision-making process.

▸ Third, choosing between manifold alternatives. Besides the objectives of EBM, assessing individual alternatives involves complex social choices and trade-offs (i.e., short-term opportunity costs vs. long-term benefits; reduced pressures and lower provision of commercial services vs. enhanced security, reduced risk, better adaptation prospects, etc.).

▸ Fourth, taking advantage of the array of different opportunities linked to EBM. While traditional measures (such as flood prevention infrastructures) are designed to respond to a particular problem, EBM approaches are linked to multiple co-benefits and may simultaneously contribute to various policy objectives such as biodiversity conservation, water quality and quantity, public health, flood and drought risk reduction, climate change adaptation, energy savings etc. Their advantages as compared to traditional approaches rely on the actual opportunity to seize the benefits of synergies or simultaneous advances across different policy and biophysical realms. However, current methodologies such as single-purpose cost-effectiveness or optimisation models might be blind to EBM co-benefits. Additionally, advantages of EBM may remain hidden in the institutional silos where sectoral policies are currently designed.

2.4 Characterising drivers and pressures affecting aquatic ecosystems

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2.4.1 Introduction

Drivers are the result of deliberate human decisions[18]. Behind any decision to obtain living species and materials such as water or minerals or to transform the energy from tides or from freshwater flows, there is the desire to satisfy a particular and well-defined demand of inputs for the production or consumption of goods and services as varied as food, manufactured products, power or recreation. The same holds true for goods and services for which production requires the modification of the natural environment at different scales, such as dredging a riverbed to improve navigation, the impoundment and diversion of water to match demand and supply or to provide security against floods, or the alteration of coastal areas to make room for population settlements and harbour facilities. Beneath any pressure there is a driver represented by the demand of goods and services provided by nature.

Drivers, or demands for goods and services provided by nature, are in general linked to multiple activities that must be coordinated in order to align individual actions with the overall objectives of sustainability. For instance, freshwater ecosystems provide water for households, agriculture, power generation and virtually all industries, as well as to maintain ecological flows. Equally, water quality is the compounded result of the demand of point and diffuse quality regulation services coming from almost all economic agents throughout space and the security against floods and droughts resulting from natural runoff regulation services spread all over social and economic agents.

The analysis of drivers requires a comprehensive analysis of the social and economic system that may include a wide range of activities including those that are actually responsible for current pressures but also the least economically relevant activities that may result in cumulative degradation processes in the future and the activities, which are not actually relevant, but might benefit from a better conservation state of the biophysical system.

Most applied analyses to date focus on few individual activities explaining single pressures with few interactions (if any) among them. Besides individual activities causing direct pressures, a comprehensive analysis requires taking into account other activities that despite

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[18] Note that the term *drivers* is used exclusively for naming the demand of nature provided goods and services (e.g. ecosystem services and abiotic outputs). To explain the underlying factors of these drivers we use terms such as determining factors, causes, etc. This way the effective drivers of pressures over ecosystems are clearly differentiated to the drivers of the drivers themselves (i.e. distinctions between indirect and direct drivers become irrelevant in this context).

their marginal economic importance may result in indirect yet significant pressures (such as maritime trade, which may result in the dispersion of invasive species or all the use and disposal of plastics that results in disturbance of marine food chains). Additionally, attention must be paid to activities that might not be currently important but only as the negative consequence of past decisions (such as foregone opportunities in tourism and sustainable agriculture due to widespread land-use changes).

Thus, drivers are easier to identify, inventory and measure than to understand. Indeed, many drivers can be measured through the observed use of particular services, such as the volume of water diverted for irrigation, the amount and composition of effluents from businesses and households or the tons of fish caught and landed per unit of time at a given place. All these numbers are being reported at local and global scales with increasing precision and detail. However, EBM and the management of the services they provide demand changing the underlying decisions that lead to these numbers.

Besides measuring, science may support policy in explaining the current use of water. For instance, the amount of demanded water may be determined by factors like the market prices of crops, the choice of irrigation techniques or the crop mix, subsidies and regulations in place, etc. A minimum understanding of the factors that determine the use of water services thus becomes essential so that focused interventions can be designed and implemented to reduce pressures, by reducing the demand of services and improving sustainability.

In other words, analysing drivers is equivalent to understanding what the demands of natural goods and services are, and how these demands are satisfied within the range of technical alternatives bounded by legal and governmental institutions and within market conditions in place. Understanding drivers is then equivalent to understanding the individual and collective decisions that result in a certain demand of services provided by biophysical ecosystems (including ESS[19] mediated by biotic processes and abiotic goods and services).

The analysis of drivers involves all the complexity of modern social systems. On one side, at a macro level, the scale of the demand of materials and energy depends on global factors such as population, income growth and climate change. At global and regional extents these processes provide a rough approximation to the scale of environmental challenges. However, the observed use of the services of natural capital is not just the result of these high-level

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[19] As expressed in the AQUACROSS concept the precise definition of nature provided services is far from a settled issue. Significant differences do exist between the concepts used at both sides of the Atlantic (see e.g. the CICES – Haines-Young and Potschin, 2012–, and the US-EPA classifications –Landers and Nahlik, 2013–). The US Environmental Protection Agency has developed a comprehensive effort to define and classify all nature based services to humans in general and of aquatic ecosystems in particular (see Landers and Nahlik, 2013). In order to encompass all goods and services provided by biophysical systems that are relevant for human welfare and thus for the social-ecological system, we include under these services all those that are mediated by biological processes (defined as the ecosystem services in the CICES classification; see: Haines-Young and Potschin, 2012), and all those that are not. All this without excluding services that are not included in the CICES classification that may be economically relevant, such as navigation, a service which provision is in the basis of relevant hydro morphological changes in European rivers and coastal areas and that result in relevant pressures over freshwater and marine ecosystems such as oil spill, spread of invasive species, noise, etc.

factors, and many other positive and negative social processes need to be factored into the analysis. This is the case for all adaptive social processes such as research, innovation, institutional development, social awareness for consequences of nature degradation and many other adaptive responses taking place at regional and local scales.

Therefore, drivers must be understood at the adequate spatial and temporal scales (see Section 2.6.3). Whilst important to understand global trends, at the regional and local level considered in the AQUACROSS Case Studies, population growth is less informative about demographic pressures than, for example, land planning and land use patterns. Similarly, innovation is less useful to understand local decisions than the prevailing incentives to implement better irrigation techniques or more selective fishing gears.

Understanding drivers of pressures therefore calls for the analysis of decisions at different levels. Decision-making processes in the global society are complex and involve multiple scales, from global to local, and multiple agents closely connected to each other. For this reason, it is important to distinguish between the different levels influencing drivers behind pressures of aquatic ecosystems. High-level variables act at regional, national or global scales and include macro processes such as demographic trends, economic growth, climate change, technological development, etc., which are then determining factors of the demand of many ESS and abiotic outputs.

The different levels of decision-making are also related to the difference between exogenous and endogenous variables at each level (Rounsevell et al., 2010). For instance, at a local level, the size of the population and the prices of inputs and outputs in fishing and agricultural markets are exogenous variables that cannot be managed by local agents. Yet, to some extent these factors can be controlled at regional levels, where land planning decisions are made, and at national or EU levels where quotas, subsidies, taxes and other decisions are taken.

Decision variables (such as fishing quotas, irrigation techniques, land use patterns, etc.), are specific to each scale and decision-making framework. Local actors can make crop and fishing decisions but have little or no control at all over market prices, technology development or climate. At the same time, at any decision-making scale, some drivers may be exogenous in the short term (i.e., technology options, land availability, drought risk, etc.) but subject to change in the longer term, as a result of spontaneous or policy driven adaptation processes.

Drivers and pressures are increasingly shaped by the extension of the progressive and cumulative impacts of human activities over marine, coastal and freshwater ecosystems as well as by the consequences of climate change and the need to adapt business and social responses to new social, political and environmental situations. Technological development and innovation processes are increasingly driven by the need to adapt to a more constrained and more uncertain supply of ESS and abiotic outputs. However, they provide the opportunity to take advantage of new business models that result from all the above-mentioned factors.

Many marine-, coastal- and freshwater-based economic activities are constrained by further deterioration of aquatic ecosystems. These ecosystems are the source of provisioning and regulating services that are essential for human life, the maintenance of many economic activities and for aquatic ecosystems themselves. Contemporary trends in agriculture, urban development, energy and transport are progressively shaped by resource scarcity concerns. For instance, scarcity and insecurity of supply explain innovation trends. They are the primary reasons to deploy old and new methods to enhance efficiency into practice with which all services provided by aquatic ecosystems are used.

Furthermore, implications of climate change have to be considered. Climate, oceans and the hydrological cycle are interlinked, and therefore also determine the availability of provisioning and regulating services. The anticipated change of climatic patterns will have considerable consequences for both, the ecological as well as the socio-economic system and thus for the provisioning of ESS and abiotic outputs. Complex decision processes that include the autonomous outcome of markets but also the regulating capacity of the institutions in place mediate both demands and technologies.

ESS and abiotic outputs provided to the social system are co-produced by humans and nature. Their provision is organised in **primary activities** such as water diversion, impoundment, extraction, wastewater management, gravel extraction, fishing, building and operation of harbours, dredging, etc. Their common purpose consists in combining natural resources with human capital and effort in order to co-produce nature-based goods and services demanded by the social system. These primary activities provide basic inputs to many, eventually all activities devoted to the **production of final goods and services** that are directly relevant for human welfare such as food, shelter, energy, recreation, security, health, etc.

Summing up, changes in ecosystems are driven by the demand of services provided by nature, including ESS and abiotic outputs. The co-production of these services is organised in primary activities that produce and convey basic inputs to the production of goods and services that are directly relevant for human welfare.

As per **pressures**, most studies to date attempt to deal with how a single pressure, such as pollution discharge, may cause a change in the ecosystem state, such as nutrient enrichment (e.g., Dahm et al., 2013; Phillips, 2014), or selective mortality of fishes (e.g., Pauly et al., 2002). More recently, attempts have been made to consider multiple pressures and their cumulative or interacting effects on ecosystem state (e.g., Matthaei et al., 2010; Ormerod et al., 2010; Piggott et al., 2012; Halpern et al., 2008, 2015; Micheli et al., 2013). Cumulative effects can exhibit additive, synergistic or antagonistic responses (Crain et al., 2008). For example, on a pan-European scale only 40% of combined stressors had an additive effect on fish in rivers whereas 60% had a synergistic or antagonistic effect (Schinegger et al., 2016). The effect of multiple stressors can also manifest differently in different aquatic realms and different components of aquatic systems also respond in different ways to conditions with multiple sources of stress (Nõges et al., 2016).

Moreover, pressures can trigger adaptive processes in ecosystems that might take them to alternative stability domains (resulting, for example, in habitat changes and/or losses, Scheffer et al. 2001): These may lead in turn to cumulative changes (e.g., triggered by "slow" and "fast" variables, Carpenter and Turner, 2000) in the structure, abundance and composition of species (leading to extirpation or extinctions as well as to the proliferation of opportunistic species or pathogens); affecting ecosystems at local, regional and global scales (from single habitats to climate change, e.g., Opdam and Wascher, 2004). Further, these may have differentiated effects over time (with threats over future ecosystems' resilience, loss of options, irreversible changes and negative legacy effects e.g., Waylen et al. 2015). Any change in ecosystem state can itself change the supply of ESS and abiotic outputs due to the inherent links between ecosystem structures (including biodiversity), functions and services (see Section 2.4.5 and Section 2.5).

Conceptual approaches have evolved to the now frequently used DPSIR (Driver-Pressure-State-Impact-Response) framework (see Cooper, 2013 and Patrício et al., 2016a for a summary of work in this area). DPSIR is a widely used approach to identify links between drivers and pressures and ecosystem state (as well as their impacts and related responses that are not addressed here). It provides a structure for the investigation of how pressures can lead to changes in ecosystem state and impacts on human wellbeing. DPSIR formalises the relationships between various sectors of human activity and the environment as chains of links. However, DPSIR models have rather favoured impact mitigation strategies and might fail to initiate structural responses as implied by EBM such as feedback loops or impact of multiple stressors (Gómez et al., 2016). Even though there is no reason why DPSIR frameworks respectively formalised relationships between drivers, pressures and states cannot be an integral part of the wider AQUACROSS Architecture, where feedback loops and multiple pressures are factored in.

In the remainder of this chapter, we focus on *how* relationships between drivers, pressures and ecosystem state can be explored as part of the AQUACROSS AF. First, in Section 2.4.2, we briefly introduce the relevant conceptual approaches (policy typologies, broad classifications, linkage matrices) that can be used to help frame this exploration. In Section 2.4.3, we consider what the role of indicators is in exploring relationships between drivers, pressures and ecosystem state, also introducing some of the key sources of effective indicators for these kinds of approaches in aquatic systems. In Section 2.4.4 we introduce qualitative and quantitative methods that can be used for exploring relationships on the demand side. Finally, in Section 2.4.5, we describe how change in ecosystem state (caused by human activities and their pressures) can affect the supply of ESS (hence linking to Section 2.5), and also how the effects of impacts on society caused by the changing nature of the ecosystem and availability of its services (driven by the demand on those services) is itself changing the nature of societal processes; this illustrates that we should also consider feedbacks from drivers and pressures into the supply-side of the AQUACROSS AF.

2.4.2 Conceptual frameworks for exploring relationships between social processes, drivers, pressures and ecosystem state

The AQUACROSS Architecture and Heuristics (below) provides a comprehensive conceptual framework to integrate knowledge, data indicators, models and other analytical tools in a meaningful way that can be taken by stakeholders and inform policy-making (Gómez et al., 2016).

Figure 7: The AQUACROSS Heuristics
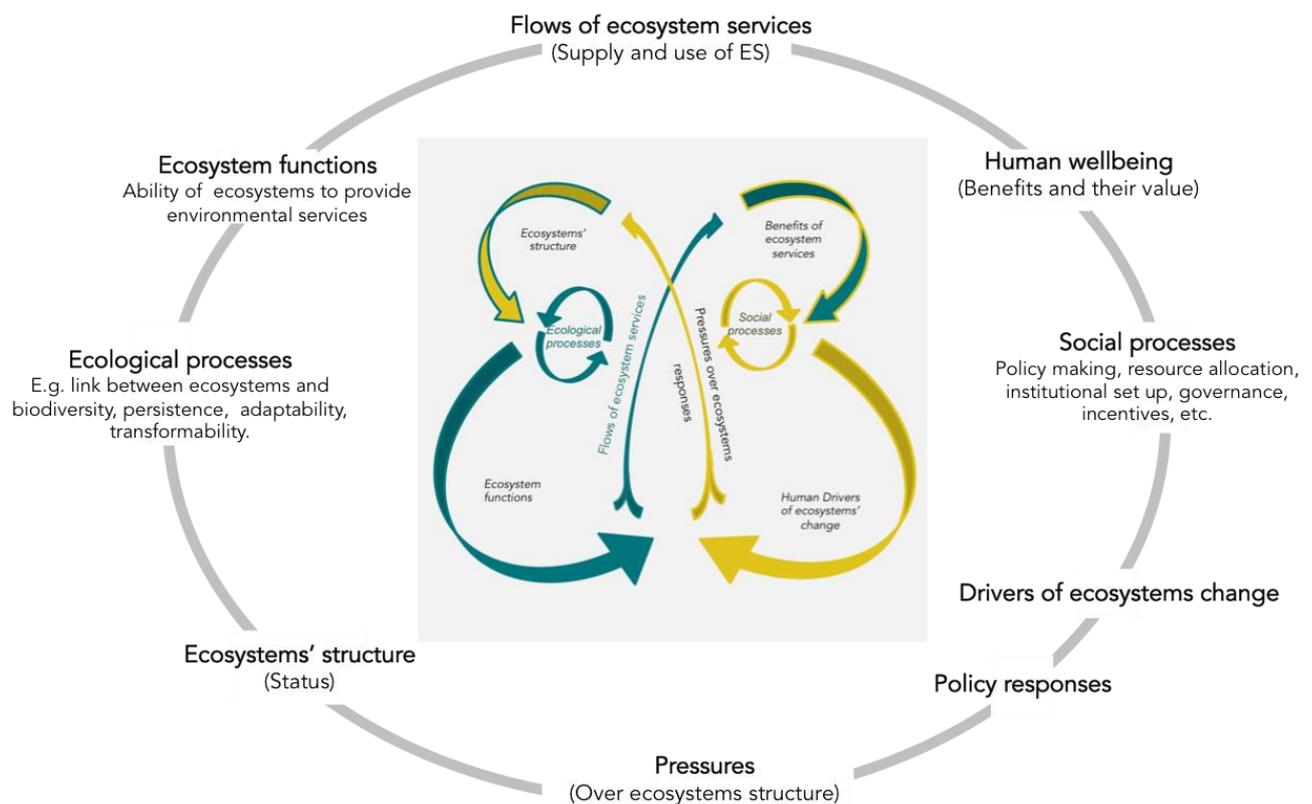
Source: Gómez et al., 2016

Information layers and flows: It shows the main information- indicator layers that need to be considered to represent the social ecological system considering the demand side (in red) and the supply side analysis (in green). Linking these layers requires empirical and theoretical models able to provide explanations. The chain of drivers, pressures, structure covers lower right part of the diagram and can be linked with the analysis of social processes (in order to explain the drivers) and to the ecosystems functions and processes (in order to understand their impact).

Besides the understanding of the economic, institutional and social processes that explain the drivers behind the pressures of aquatic ecosystems the AF requires a clear definition of each component of D-P-S relationships (Böhnke-Henrichs et al., 2013; Liquete et al., 2013; Culhane et al., 2016). Therefore the DPSIR framework is a practical way to develop this part of the demand side component within the AQUACROSS Architecture.

As a starting point, the relationships between drivers, pressures and ecosystem states are often described using relational chains within a linkage-based framework (Rounsevell et al., 2010; Knights et al., 2013; Robinson et al., 2014). This approach involves defining the drivers, the pressures, and the ecosystem components (e.g., ecosystem state), and developing typologies or comprehensive lists of all relevant elements, which can be linked in those relational chains (for example, see Figure 7).

We link the socio-economic and the ecological systems by making a clear distinction between:

▸ The activities that benefit from the provision of natural goods and services for the production of final goods and services that are of direct concern for human welfare (such as food, shelter and recreation).

▸ The drivers of pressures affecting ecosystems, represented by the specific demands of naturally provided goods and services (water, fish, energy, materials, regulation services, etc.) in the quantity, quality required at specific places and moments of time.

▸ The primary activities that (co-) produce goods and services provided by natural capital (such as extraction of water, mining, fishing, navigation, dredging, building and operation of harbours, pollution, etc. [20]) that are of direct concern to explain the pressures over ecosystems.

From a social perspective, to understand the demand of primary inputs provided by nature (the drivers), we must understand the demand and supply of final goods and services to which these inputs contribute. In turn, to understand the pressures and the subsequent changes of ecosystem components, we should focus on the primary activities that are directly interacting with the ecosystem and cause a pressure so that we know where the pressure is and with what intensity. This helps determine how exposed the ecosystem is (a key aspect in assessing sensitivity and the effect of drivers and pressures acting on the ecosystem).

This identification of the different types of activities related to a particular driver or several drivers is new in terms of how we think about the DPSIR-type approach. It is important to clearly distinguish between these different parts within the relational chain, and have a common understanding of the categories in order to develop comparable outcomes of the

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[20] Following the standard definitions of economic activities use "production of final goods and services" for the first kind of activities (encompassing the so-called secondary and tertiary sectors of the economy) and "primary activities" for the second (encompassing the production of all inputs such as energy, materials and other services that are essential to the production of final goods and services). See the EU Classification of Economic Activities: http://eur-lex.europa.eu/legal-content/EN/TXT/HTML/?uri=CELEX:32006R1893&from=EN

relationships across geographical regions and/or across aquatic realms, regardless of the drivers, types of activities, pressures or ecosystem changes, which may occur (Cooper, 2013). By identifying clear and consistent relationships, this facilitates the identification and selection of indicators and the focus of management options in a coherent way (Rogers and Greenaway, 2005) that can be further used to analyse the causal links between biodiversity, ecosystem functions and services (see Section 2.5).

Figure 8: Example of a relational chain
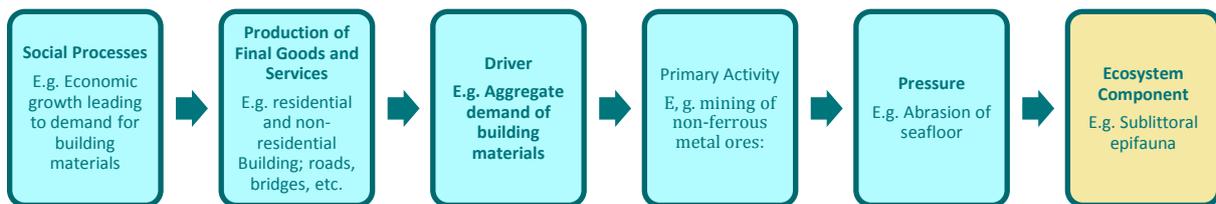
Legend: Example of a chain from the social processes that determine the production of final goods and services that explain the demand of ecosystem goods and services (*Drivers*) and the activities addressed to meet this demand (*Primary activities*) that result in specific pressures over particular ecosystems' components. While traditional management focus on the primary activities that link drivers to pressures, the EBM promoted by AQUACROSS aims at being an integrated management response along the whole linkage framework.

Source: Pletterbauer et al. (2017)

Typologies of final and primary activities, drivers, pressures and ecosystem components can be developed such that they are clearly linked to policies relevant for aquatic ecosystems. A starting point must be the EU classification of economic activities (EU-NACE).

For example, policies such as the MSFD or the WFD already list a number of pressures (e.g., selective extraction of species) and ecosystem components (e.g., fish) and these can be included directly in typologies, or can be indirectly linked to particular elements in typologies. Through the relational chains, the human activities responsible for pressures and change in ecosystem state can be identified (e.g., fishing). Deliverable 4.1 of AQUACROSS (Pletterbauer et al., 2016), reviews all relevant typologies for aquatic realms and develops an overarching classification that aligns nomenclatures and definitions of drivers and pressures from the Habitats Directive, Marine Strategy Framework Directive and the Water Framework Directive. The classification is organised into a linkage framework where linkages between drivers, pressures and aquatic ecosystem components (e.g., fish, invertebrates, habitat type) are identified using expert knowledge and evidence from literature. This can be used to help frame and contextualise the analyses in case studies going forward.

2.4.3 Indicators of drivers, pressures and ecosystem components/state

The linkage framework integrated into the AQUACROSS Heuristics and Architecture facilitates the identification of indicators needed to describe the system from the demand side. Generally, an indicator provides aggregated information on target criteria (Wiggering and Müller, 2004), and tries to depict qualities, quantities, states or interactions that are not directly accessible (Kandziora et al., 2013).

A clear and common understanding of the concepts of indicators, indices and metrics is required. In AQUACROSS we will consider the following definitions:

▸ **Indicators** – *variables that provide aggregated information on certain phenomena, acting as a communication tool that facilitates a simplification of a complex process. It relates to the component or process responsive to changes in the social-ecological system, but does not necessarily have a measurable dimension, and therefore it is not an operational tool in itself.*

▸ **Indices** – *metrics whose final outcome should be easily interpreted by a non-specialist within a qualitative continuum. It can be a quantitative or qualitative expression of a specific component or process, to which it is possible to associate targets and to identify trends, and which can be mapped. It is how an indicator becomes an operational tool used within management, regulatory or policy context.*

▸ **Metrics** – *quantitative, measured, calculated or composite measurements based upon two or more measurements that help to put a variable in relation to one or more other dimensions.*

In order to populate the demand side of the analysis with indicators this might require metrics of activities, drivers, pressures, components, etc. Deliverable 4.1 (Pletterbauer et al., 2016) explores the availability of these across different aquatic realms such as the indicators used within the water framework directive reviewed by Birk et al. (2012) or the indicators used within the Marine Strategy Framework Directive (EC, 2011). We also consider how or where they can be used to evaluate change in the SES rather than just describing state.

The purpose of an assessment strongly determines the type of metric or index needed to address a problem and the spatial scale of application (Feld et al., 2009). There has to be a clear representation of the *indicandum*, a proven cause-effect relationship, an optimal sensitivity of the representation, information on adequate spatio-temporal scales, transparency including a reproducible methodology, a high degree of validity and representativeness of the available data sources, an optimal degree of aggregation, information and estimations of the normative loadings, high political relevance, high comprehensibility and public transparency, relations and responsiveness to management actions, an orientation towards environmental targets, a satisfying measurability, a high degree of data availability, a high utility for early warning purposes (Wiggering and Müller,

2004 in Kandziora et al., 2013). Finally, trade-offs between costs and effectiveness is a crucial factor.

We note that indices and metrics usually can only represent individual parts of the framework at a time (e.g., the state of benthic invertebrate communities or the size of a particular activity), and this limits their potential for evaluating causal relationships in the system. We also note that metrics can only be used in data rich situations where metrics of different parts of the framework (driver, pressure and ecosystem state) can be calculated and linked using quantitative approaches (see Section 2.4.4). However, the use of indicators is limited in situations where data is limited, such as in European regional seas. In these cases, qualitative approaches can be used, starting from linkages, to make an assessment where relational links are inferred but not quantitatively measured (see Section 2.4.4).

2.4.4 Methods to analyse links between drivers, pressures and ecosystem state

The information layers described in the previous section, and the standardised and consistent information systems they conform, facilitate the description and the assessment of each one of the building blocks of the AQUACROSS Architecture. Going one step further requires being able to build upon the links between one layer and the other as well as within the social and ecological systems themselves. This is the role of qualitative and quantitative analytical models (see below). Beyond description and assessment, the distinctive character of these models lies on the fact that they allow navigating through different information layers and building comprehensive scenarios, storylines, assessments of the overall social-ecological system and the development of comprehensive decision tools and platforms to support the identification, design, implementation and assessment of EBM options.

Figure 9: Analytical models involved in AQUACROSS

From pressures (and responses) to ecosystems' structures

To explain impacts over ecosystems' structures of pressures (and responses outcomes in terms of improved ecosystems structures)
Impact assessment models, etc.

Ecological processes

Analyse ecological processes with emphasis in the link between biodiversity and ecosystems. Based upon the status of ecosystems' structure

From ecosystem functions to ecosystem services

Assessment of the potential of ecosystems to provide a portfolio of services

From ecosystem services to ecosystem benefits

Assess social (economic and policy) ability to put ecosystems services into value for human well-being.
Resource efficiency models, water food energy, climate change integrated models, etc.

Social processes

Assessing the status of the economy, policy making institutions and explain the main outcomes of social processes: drivers and responses.
Economic models: macro and micro. General Equilibrium, I/O, demand models. Cost benefit analysis.
Risk assessment models. Decision support platforms.

From drivers to pressures and responses

To explain how drivers are linked to specific pressures over ecosystems
Demographic models, agronomic models, land use models, agent based models, etc.

Flows of ecosystem services
(Supply and use of ES)

Ecosystem functions
Ability of ecosystems to provide environmental services

Ecological processes
E.g. link between ecosystems and biodiversity, persistence, adaptability, transformability.

Ecosystems' structure
(Status)

Human wellbeing
(Benefits and their value)

Social processes
Policy making, resource allocation, institutional set up, governance, incentives, etc.

Drivers of ecosystems change

Policy responses

Pressures
(Over ecosystems structure)

AQUACROSS Models

What links? What for? *Examples*

Source: AQUACROSS Concept (D3.1)

Legend: The analytical models involved in AQUACROSS are purposely designed to mobilise existing knowledge and provide the best possible explanation of the links involved (Gómez et al., 2016).

Linking drivers with pressures and ecosystem structures

Effective resource management will require the targeted selection of analysis method that accurately predicts the outcome of possible management decisions or future scenarios. That involves assessing the drivers and pressures in relation to the ecosystem state of a system and making educated decisions about the response of that state to changes. Various qualitative and quantitative tools for D-P-S assessment are available and widely used across different aquatic realms, which are reviewed in detail in Deliverable 4.1 (Pletterbauer et al., 2016). There are various approaches that deal with the analysis of linkages between drivers, pressures and ecosystem state along a gradient of qualitative to quantitative applications.

First **qualitative** or semi-quantitative approaches can be used to gain greater overall understanding of social-ecological systems using mainly expert judgement and outcomes from different empirical studies or literature reviews as a basis (e.g., Knights et al., 2013; Halpern et al., 2008, 2015). The linkage framework can be further developed with approaches, which do not directly rely on 'measured quantifications', such as fuzzy cognitive mapping (FCM, see Lorenz et al., 2016b) or Bayesian belief networks (BBNs, see Aguilera et al. 2011). They are able to integrate causal knowledge and to investigate complex systems. However, different authors (e.g., Mouton et al., 2009; Boets et al., 2015; Hamilton et al., 2015) concluded that expert models can only be successfully applied when there is already detailed information on the ecology and response to environmental parameters available for the respective system and model evaluation is seen as critical for developing rigorous expert models (e.g., Chen and Pollino, 2012; Hamilton et al., 2015).

On the other end of the gradient a broad variety of **quantitative** correlative models exist that focus on the causal relationships of ecological components, ecosystem conditions and/or human activities in high detail but in a narrower view based on empirical data. Correlative approaches significantly differ in their performance with respect to overall predictive performance, generality and transferability, their causal interpretability with respect to relevant background theory or graphical representation for communication in an open management process (for details see Deliverable 4.1: Pletterbauer et al., 2016).

Also process-based models are widely used including, hydrological models or catchment models for nutrients (e.g., Venohr et al., 2011) that in turn can be used as input for e.g. species distribution modelling (Dormann et al., 2012). A process-based model is the mathematical representation (formulated as mathematical functions) of one or several processes, including physical- or biochemical-based processes, based on a function of generic principles or empirical knowledge (expert knowledge) and might be fitted on the basis of empirical data.

In process-based models causality is defined *ex ante*, assuming that the model structure and process formulation are correct whereas in correlative methods mainly *post hoc* interpretation is causal besides the fact that also the explanatory variables are employed in such a way that they are expected to represent causal mechanisms. In that sense causality is not necessarily assured and a critical issue in both approaches (Dormann et al., 2012) and

there is strong evidence that combined model development (use of different models and tools for the same analytical problem) is of help. On the one hand, it improves the overall performance (Araujo and New, 2007); on the other, it leads to more robust models (Hamilton et al. 2015) with higher interpretability or ecological importance (e.g., Boets et al., 2015; Dormann et al., 2012), Hence, this also contributes to better communication in open planning processes.

Understanding the social processes that explain the drivers of pressures: activities, institutions, and ecological constraints

On descriptive grounds, the assessment of drivers of ecosystem change must provide the elements to screen out the multiple ways societies trigger changes in nature. Emphasis must be placed on those social processes that result in significant ecosystem changes, and then in shifts in the ecosystems' structure and dynamics, and particularly in those drivers that push the system beyond its sustainability thresholds. Comprehensive lists and detailed classifications of drivers might help for this screening exercise and therefore are useful to focus on relevant drivers as well as to avoid omitting potentially relevant interactions.

On analytical grounds, the assessment of drivers of ecosystem change must be designed in such a way so as to provide the best possible understanding of social choices, both about the demand of relevant ESS and about the technological alternatives chosen to meet those demands (which in turn determine the pressures stemming from the satisfaction of the demand of ESS). However, looking just from the perspective of the demands for ESS and abiotic outputs could result in overlooking some pressures. These might be unintentional or caused by activities which are not directly a demand of goods and services: for example, diffuse pollution from activities removed from the study site. Therefore, different perspectives should be considered, as well as considering activities related to the demands on goods and services. It is also possible to start by looking at what pressures and specific activities are relevant to the system.

Different approaches for analysing the interactions in the demand-side relationship exist that provide the possibility to analyse and explore linkages between economic activities, drivers, pressures and ecosystem components. Drivers of ecosystems change are the main outcome of social processes (see the AQUACROSS innovative concept). All pressures are driven by economic activities such as agriculture, transport of goods and services, land occupation and development, fishing, tourism and recreation, etc. that demand ESS of different kinds (such as water for irrigation, navigation services, room for houses, provision of fish, landscapes, etc.). All these activities can be classified and described by using common macroeconomic accounting methods (such as those used in national or regional accounts). In addition, their importance for human welfare can be approached by using standard economic concepts like value added, employment, input output coefficients, etc.

A first basic step to investigate the drivers of pressures consists in identifying the set of goods and services enjoyed at the case study level. That is to say making the inventory of the current, past and prospective use of ecosystems services at the level of the case study. This

exercise serves to scan what the benefits are and who the beneficiaries are, sometimes with high level of precision. For instance farmers, local manufactures, anglers, etc. can be singled out as the direct beneficiaries of rival and excludable services such as water for irrigation, raw materials, recreation, food, etc. Similarly collective benefits of public services such as water security, flood and pest control may be identified.

The identification of benefits and beneficiares leads to the identification of economic activities (such as farming, manufacturing, fishing, recreation, etc.) thus facilitating the understanding of drivers of demand for ESS. Analysing these activities (their added value, the job opportunities, the use of provisioning ESS – such as freshwater water, wild fish, navigation, etc. – their resource efficiency, the regulations that allow or restrain the use of water related ecosystems' services, etc.) is key to understand and explain the demand of provisioning ESS and then the underlying drivers of ecosystem change.

This activity-based approach allows focusing on individual ESS (such as provision of water for irrigation, power generation, navigation, etc. or other particular services such as agrochemicals disposal, recreation, runoff regulation, etc.), for which the demand can be linked to the size and the characteristics of the sector. In fact, the analysis of economic activities is the basis to understanding the current and prospective demand of aquatic ecosystems services that drive ecosystem change.

Despite its importance to explain many drivers of ecosystem change, the analysis of economic activities is not enough to get a full understanding of the demand of ESS. This is because activity-based (or sectoral) analysis does not account for non-market and non-monetary services, ignores interactions between economic activities that result in emerging drivers, and tends to place emphasis on the scale of the sector. All drivers of ecosystem change must be properly understood at different time and spatial scales. Non-market ESS help understand the opportunity costs of matching past, current and prospective demands of provisioning ESS to the different economic activities. Furthermore, those services provide the background to understand critical trade-offs linked to business-as-usual scenarios and to develop strategies for a sustainable future.

Besides activities, institutions and governance systems play a central role in how humans relate with nature (Ostrom, 1990; Lowry et al., 2005; Abunge et al., 2013). They form part of the determining factors of the demand of ESS of all economic activities. Institutions encompass social rules and interactions between social agents, determine property rights, power structures, incentives, access and control of natural resources and shape individual and collective decisions in many ways. Examples of institutions are property rights systems (common property, open access, quotas, fishing allowances, water use rights, etc.), social norms and rules (such as agreements to protect marine and terrestrial reserves, national and international treaties, etc.), economic policies (land development plans, river basin management plans, energy, agricultural, fisheries policies, etc.).

Drivers of ecosystem change are outcomes of social processes. Thus, besides the description and measurement of activities and associated demands of ESS, the analysis of drivers

requires a proper understanding of the determining factor of both the size of the economic activity and its use of ESS.

The first kind of analysis refers to understanding each economic activity itself. Explaining the production level of agriculture or fishing activities requires a proper understanding of the demand for food, the size of the population as well as their incomes and the public policies in place (irrigation development plans, agricultural policy, etc.) and their evolution throughout time (population and income growth, technological innovation, etc.)[21]. Population growth is linked to the scale of demand of ESS, both at global and local scales, and drives the expansion in the demand of food, energy, and land. This, compounded with trends in economic development, trigger social trends such as land transformation from forest to agriculture, urbanization, resources depletion, infrastructure development, etc. (Dasgupta and Ehrlich, 2013). Global economic growth and demography are the main drivers of the overall consumption of final goods and subsequently of the demand of natural resources and other ESS (MA, 2005; Gomez-Baggethun and Ruiz-Perez, 2011; IPCC, 2014).

The second kind of analysis refers to understanding how a particular economic activity results in the effective demand and use of a given quantity and quality of ESS. For instance, the demand for water associated with the production of the same amount of wheat depends on many factors such as soil type, weather, the irrigation system in place and the water source used. Likewise, the effluents produced by any urban settlement not only depend on the number of people but on the status of the sewage system, the wastewater collection and treatment system in place as well as the costs and prices and on the way water is finally discharged to the environment.

The previous distinction is essential to understand the factors determining the demand of ESS, and then to identify challenges and opportunities to improve the management of ecosystem demand in such a way that leads to more sustainable pathways of concurrently satisfying human needs and improving the status of ecosystems.

Though the analysis of drivers might be seen as following a one-way pathway from the social system to the ecological system it is also a critical piece of a holistic approach (built over the entire AQUACROSS Architecture). This is particularly important when considering the determining factors explaining socio-economic activities that drive ecosystem change. These activities are increasingly shaped by the need to adapt to changes, most of them detrimental, in the ecological system such as climate change, higher and more severe risks of various kinds, scarcity, depletion of critical assets, etc. These mutual adaptation processes are important nowadays to understand trends in activities as crucial as transport, agriculture, power production, fishing, etc. (Perez, 2004; Carpenter et al., 2006). The selection of production techniques (e.g., drip instead of gravity in irrigation, aquaculture instead of wild

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[21] Sometimes these determining factors are called high-level drivers as different from low-level drivers. See, for instance, Knights et al., 2013. These definitions are confusing and force distinguishing drivers that are exogenous and endogenous to ecosystems. We prefer making a clear distinction between the factors that determine the economic activities on one side and the drivers, or factors that drive ecosystems change.

fisheries, electrical instead of fuel powered cars) and many other decisions can only be explained as responses to a more resource constrained environment (Dietrich et al., 2014). In a similar sense, practices such as sustainable urban drainage, soil conservation measures, smart transport networks etc. can only be considered as part of adaptation strategies to reduced risks and adapt to climate change.

The social drivers of ecosystem change are increasingly wrought by the extension of the progressive and cumulative impacts of human activities over marine, coastal and freshwater ecosystems as well as by the consequences of climate change and the need to adapt business and social responses to new situations. Technological development and innovation processes are increasingly driven by the need to adapt to a more constrained and more uncertain supply of environmental services and by increasing opportunities to take advantage to the new business opportunities that result from all the above-mentioned factors. Many marine-, coastal- and freshwater-based economic activities are increasingly constrained by further deterioration of aquatic ecosystems. These ecosystems are the source of provisioning and regulating services that are essential for human life, the maintenance of many economic activities and for aquatic ecosystems themselves and the services they provide. New trends in activities such as agriculture, urban development, energy and transport are in the context of emerging trends in technology driven by resource scarcity concerns. Scarcity and insecurity of supply is an emerging driver of innovation. These are reasons to put into practice old and new methods to enhance the efficiency with which all services provided by water ecosystems are used. Furthermore, implications of climate change have to be considered. Climate, oceans and the hydrological cycle are interlinked and determine the availability of provisioning and regulating services. The anticipated change of climatic patterns will have considerable consequences for both the ecological as well as the socio-economic system. Further, it is important to bear in mind that complex decision processes that include the autonomous outcome of markets but also the regulating capacity of the institutions in place mediate both demands and technologies.

2.5 Understanding causal links between biodiversity, ecosystem functions, and services

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2.5.1 Working framework: from state to benefits

The analysis of drivers and pressures (see previous section) fits into the broader AQUACROSS Architecture (see above) through the consideration of the changes in ecosystem state that are affected by pressures and driven by the demand of ecosystem services (ESS). An impact on the state of the ecosystem can lead to changes in ecosystem functioning (on the ecological side) and the subsequent supply of ESS and abiotic outputs (socio-economic side). For example, changes in the supply of ESS and abiotic outputs can in turn have impacts on the demand side resulting from shifts in human activities associated with the use of those affected ESS and abiotic outputs. These changes can be related to the value of those services or through the benefits gained from those services. This section deals with the relationship between biodiversity and ESS. These links are also important when considering where the focus of management should be, as management can target human activities (drivers), pressures or ecosystem components, which in turn may affect the supply of ESS.

Within the holistic analysis of SES proposed by AQUACROSS, and considering the AQUACROSS Architecture, this chapter focuses on the supply-side perspective (Figure 10), which describes the capacity of the ecological system to deliver services to the socio-economic system, thus contributing to human welfare. In this sense, Chapter 2.5 receives inputs and builds upon the analysis presented in the previous chapter. Understanding how this change in ecosystem structure and status may result in a change in the capacity of the ecosystem to deliver services requires both the understanding of how pressures affect the state of the ecosystem and how a change in state may affect the supply of services.

The effects stemming from a wide range of pressures have been mostly analysed through field observations or experimental manipulations. These studies tend to inform us about the effects at the species or, sometimes, the process level.

Nevetherless, the challenge is to understand how or if these changes would lead to any change in the capacity of the ecosystem to supply services. However, metrics used to describe how pressures change ecosystem state may not be the appropriate ones to describe how the ecosystem contributes to the delivery of all services. For example, most studies on the effects of abrasion from trawling fishing activity describe the effects in terms of changes in abundance or sometimes biomass of benthic invertebrate species (Kaiser et al., 2006) or of aquatic submerged vegetation (Costa and Netto, 2014).

In order to consider the effect of abrasion on the service *Mediation of waste, toxics and other nuisances* (see Table 9 below), not only would we need to know about abundance and/or biomass, but we would also need to know how the different components of the benthic system (flora and fauna) can be described in terms of their role in this regulating and maintenance service. This could be through consideration of biological traits that are associated with *Mediation of waste, toxics and other nuisances* e.g. the role of different fauna species in bioturbation or the role of seagrasses in phytoremediation.

The remainder of this chapter explains our current understanding of the relationship between biodiversity and the supply of ESS.

Figure 10: Supply-side in AQUACROSS Architecture
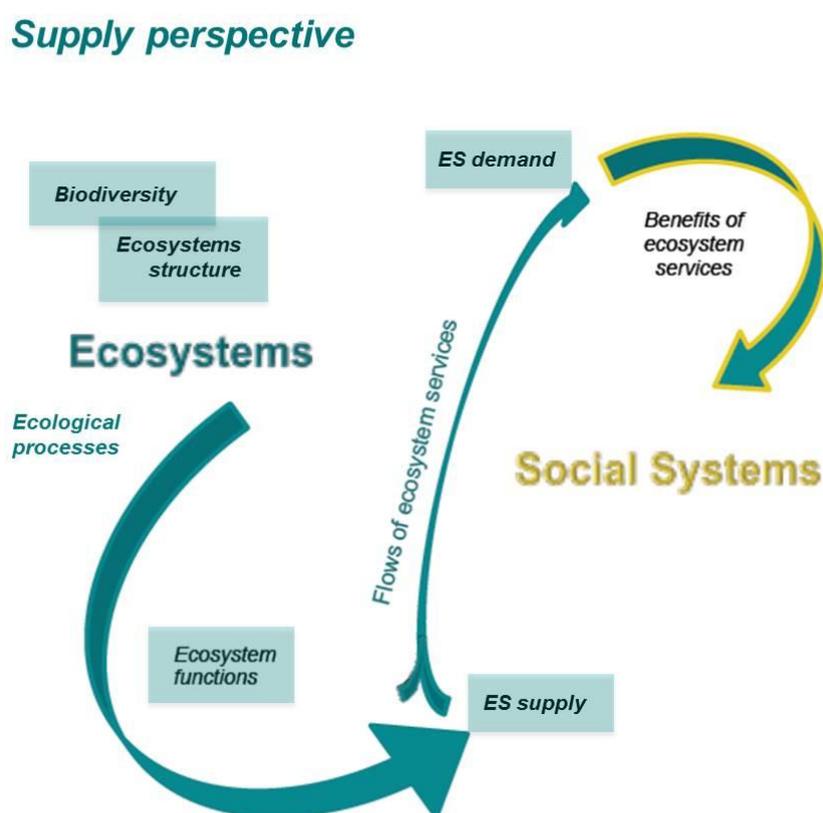
Legend: Representation of AQUACROSS Architecture (adapted from Gómez et al., 2016), highlighting the supply-side perspective addressed in this chapter.

The AQUACROSS innovative concept (Gómez et al., 2016; see Figure 3) has defined and identified the key points and links within the SES that are relevant for this stage of implementation of the AQUACROSS AF. Biodiversity (BD) (directly measured or as captured by the state of ecosystems) and the ecological processes ensuring crucial ecosystem functions (EF) that enable the supply of ecosystem services (ESS) are central themes to this stage of the

AF. Figure 11 illustrates the flows that need to be considered in order to understand the causal links between biodiversity, ecosystem functions and services.

Figure 11: Representation of a Social-ecological System
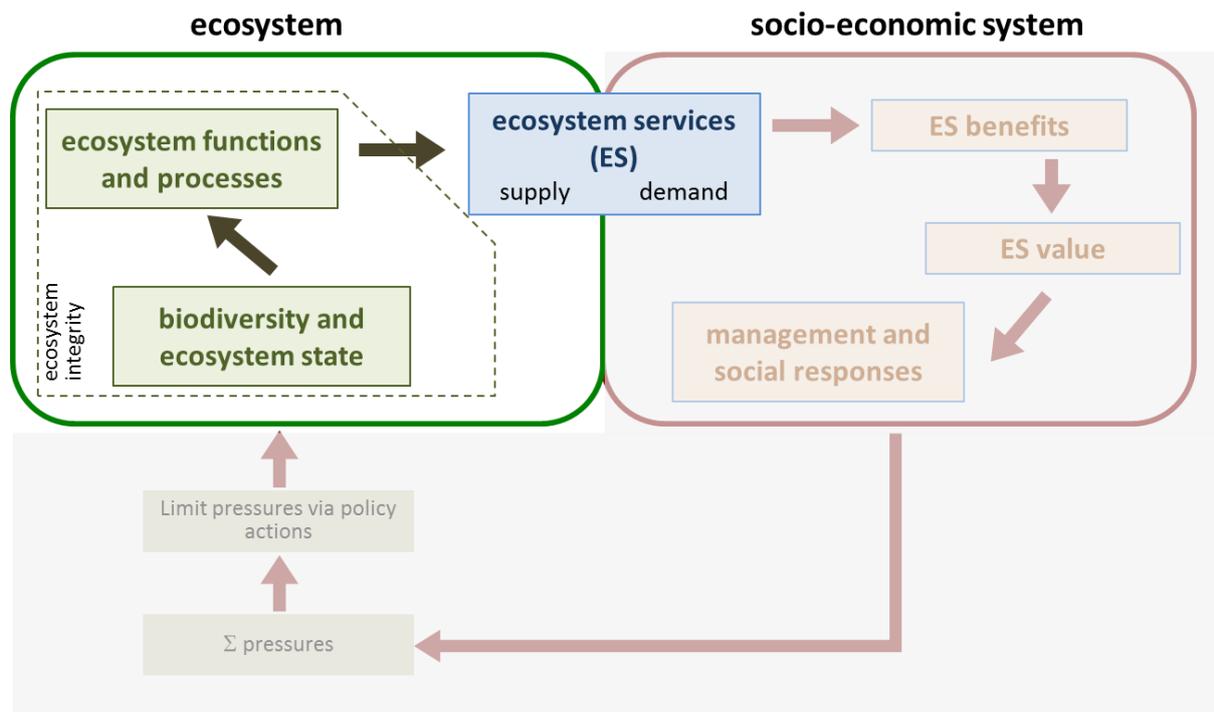
Legend: Representation of Social-ecological Systems highlighting the flows from biodiversity to ecosystem services (adapted from Liquete et al., 2016b and Haines-Young and Potschin, 2012).

Considering the AQUACROSS AF, the supply of ESS, i.e. the potential or capacity of the ecosystem to supply services, is directly linked to the ecological system (Figure 10) while the demand of ESS, i.e. whether and how the service is actually used, is the entry point to the socio-economic system (Figures 10 and 11). For example, from the supply side, an assessment could be made of the capacity of the system to supply 'Seafood' and would include the biomass of all fish and invertebrate species that can potentially be used for 'Nutrition' (i.e., the stocks). From a demand side, the flow of the 'Seafood' service to society would be the individuals that are actually taken (i.e., the catch). In this sense, a change in ecosystem state and biodiversity can lead to a change in the supply of services but not necessarily change the demand on the service.

The demand for ESS also differs from benefits to society (Figure 11). Benefits are generated by ESS in combination with other forms of capital and have a direct impact on human welfare (Fisher et al., 2008). For example, the service 'Seafood' is a provisioning service leading to the benefit of nutrition. Recognising that benefits often require a production boundary (Culhane et al., 2016; Sousa et al., 2016) can help define the differences between ESS demand and benefits, but not for all. For example, flood protection is both a service (e.g.,

through the attenuation of wave energy) and a benefit (where there is the avoidance of providing artificial flood defences and people are not affected by flooding). Depending on the service, certain individuals or groups benefit at different times. In addition, the benefits of a service can be numerous, for example, for the previous example 'Seafood' the benefits are broader than nutrition. The livelihoods of fishermen, for example, are also assumed to be captured within this service (e.g., Hattam et al., 2015). Avoidance costs should also be considered (e.g., the avoidance of disease and subsequent health costs due to regulation of pests and disease in the ecosystem) and so should wider benefits than purely economic ones (many services do not have a direct monetary value that can be measured or a monetary value which truly reflects the full benefits offered by the service). On the other hand, a benefit can also be dependent on a number of ecosystem functions and services, as for example 'Recreation and leisure'. Associated to benefits there is also the value(s) that is placed upon the benefits (Figure 11).

Several classification systems are available (Böhnke-Henrichs et al., 2013; Liquete et al., 2013; Culhane et al., 2016; Lillebø et al., 2016) to make the AQUACROSS Architecture and Heuristics operational. However, the literature shows that for each of these themes, i.e. biodiversity and ecosystem state, ecosystem functions, and ESS, definitions and classification schemes adopted vary according to specific objectives, scales of application, and often in relation to specific policies. Hence the use of common classifications is more an exception than a rule (as briefly reviewed in the following Sections 2.5.3 and 2.5.4).

The next sections explain how to move from the conceptual framework described above towards an operational framework, firstly by reviewing current knowledge on BD-EF-ESS causal relationships (Section 2.5.2), and then by identifying approaches (i.e indicators – in Section 2.5.3, and modelling approaches – in Section 2.5.4) deemed more adequate for an effective assessment of the supply-side, just preceding benefits (BD-EF-ESS – Benefits).

The work described in this chapter will provide guidance for the analyses to be performed within each case study, i.e. for evaluating the supply-side when implementing the AF. The outputs of this work will also contribute directly to data analyses (see Section 2.6), forecasting of biodiversity and ESS provisioning and providing support to facilitate and promote science/policy communication (see Section 2.1).

In a first step, a review and meta-analysis of the current state of knowledge of selected links between biodiversity, ecosystem functions and ESS have been performed (see Section 2.5.2). This meta-analysis contributes to the identification of knowledge gaps and causalities as well as weaknesses associated with existing approaches (e.g., unsuitability of existing causality models to deal with impacts of environmental stressors). The conceptual and methodological guidance introduced in this chapter to characterise the possible causal pathways between biodiversity and aquatic ecosystem functions and services, together with the meta-analysis of existing information will result in a suite of new, integrative indicators to quantify such relationships.

In addition, relevant and feasible indicators and metrics to measure changes in aquatic biodiversity, ESS status and trends, and ecosystem resilience according to Action 5 of the EU 2020 Biodiversity Strategy, will also be identified (see Section 2.5.3). The selected indicators and associated metrics will allow capturing relevant social-ecological dimensions at the case-study level, including climate change adaptation and mitigation (green/blue infrastructure, carbon sequestration), human activities supported by ESS (like tourism, maritime traffic, and fisheries), and biodiversity.

Attaining the AQUACROSS core goal of expanding current knowledge and fostering the practical application of the EBM concept for all aquatic (freshwater, coastal, and marine) ecosystems as a continuum (see Section 2.1), the importance of biodiversity at land-water interfaces (ecotones and ecoclines) in relation to drivers and pressures affecting aquatic ecosystems will also be assessed. Species richness is often relatively high in ecotones due to the proximity and functional links between the adjacent ecological systems that are combined with the processes within the ecotone itself. In these adjacent ecological systems, the genetic diversity may also be high, especially where the ecotone coincides with the extremities of species' distributions (Naiman and Décamps, 1990). In the case of transitional waters systems, between the river and the sea, their dynamic is better explained by a two-ecocline model, which represents a boundary of more gradual, progressive change (both spatial and ecological) between two systems (Attrill and Rundle, 2002). AQUACROSS will explore how ecotones/ecoclines contribute to the resilience and resistance of the associated ecosystems to various classes of human (and natural) disturbances (related with drivers and pressures identified as described in Section 2.4) through:

▸ Identification of environmental issues linked with resilience, namely how different types of biodiversity relate to resilience (see Section 2.5.2).

▸ Identification of biodiversity aspects that might promote resilience of ecotones (land-freshwater, land-marine) and of ecoclines (freshwater-marine) (see Section 2.5.2).

▸ Identification of biodiversity indicators and associated metrics suitable to forecast resilience (see Section 2.5.3).

▸ Integration of resilience in biodiversity causal links with ecosystem functions and services over the aquatic realms continuum (from catchment to sea) (see Section 2.5.4).

The role of habitats (from freshwater to marine environments) and ecotones/ecoclines (land-freshwater, land-marine, freshwater-marine) in the causal links between biodiversity, the ecosystem functions and the supply of ESSwill be assessed in general in all the case study areas, whilst some case studies will act as show cases where the proposed methodologies will be tested. This will be achieved by creating habitat-function-service matrices for the case studies at different scales (see Section 2.6), where functional and trait biodiversity will be taken into consideration. Also through spatially-explicit mapping techniques to deal with knowledge/data gaps (Maes et al., 2014) and forecasting the nature of causality links with biodiversity (meta-analysis). Since current knowledge of the links between measures of biodiversity (e.g., species richness, functional diversity) and ESS that directly affect human

well-being is still patchy, special attention will be given to how structure, diversity and dynamics of natural communities underpin their ability to deliver ESS. In this context, some of the case studies will be used as show cases by using ARIES modelling approach (see above) that deals with scattered and/or incomplete datasets and provides an assessment of uncertainty.

Information assessed at this stage will provide insight on how biodiversity-related causal links are affected during disturbance and recovery through a suit of statistical approaches (see Section 2.5.4), and how to take advantage of the ecologically valuable properties of ecotones/ecoclines to contribute to the development of management guidelines in aquatic ecosystems (see Section 2.1). Ultimately, which lessons can be learnt from the case studies will be identified and may lead to the reformulation of the original set of research questions identified in the AF or the overall concepts of AQUACROSS.

2.5.2 Literature review of links between biodiversity, ecosystem functions, and services

Over the past decades, extensive scientific research has been conducted to ascertain the link between biodiversity and ecosystem functioning (hereafter BEF) on the one side and between biodiversity and ecosystem services (BES) on the other. However, aquatic (and especially freshwater) ecosystems have received relatively little attention, and it may be disputed as to whether evidence of BEF relationships obtained from research in terrestrial ecosystems may be extrapolated to aquatic realms (e.g., Duncan et al., 2015). Subsequently, the current state of knowledge on links between BD, EF and ESS in aquatic realms (i.e., freshwater, coastal and marine) will be reviewed. This will include evaluating the state of the art regarding i) mechanisms and shape of aquatic BEF and BES relationships reported in the open literature; ii) to what extent BEF and BES relations are ecosystem-specific or whether they are interchangeable; and iii) current research limitations and needs in aquatic BEF and BES studies. This will hence aid in identifying key areas and bottlenecks in establishing aquatic BEF and BES relationships by carrying out a rule-based approach literature review, by reviewing and summarising existing scientific and non-scientific literature related BEF and BES.

To pave the way for better-integrated and more productive research in this area, and particularly for the adoption and/or development of indicators, associated metrics and models within and hopefully beyond the AQUACROSS context, it is essential to provide precise classifications and standard definitions of biodiversity and ecosystem status, ecosystem functions and ecosystem services.

Biodiversity and ecosystem state

Biodiversity has an inherent multidimensional nature, spanning genes and species, functional forms, habitats and ecosystems, as well as the variability within and between them (Gonçalves et al., 2015; Laurila-Pant et al., 2015). Often regarded as a measure of the

complexity of a biological system (Farnsworth et al., 2012, 2015), biodiversity is usually taken by an abstract ecological concept (Bartkowski et al., 2015). Since preventing the loss of biodiversity is increasingly becoming one of the important aims of environmental management, biodiversity must be understood and defined in an operational way (Laurila-Pant et al., 2015).

Farnsworth et al. (2015) have defined biodiversity as the information required to fully describe or reproduce a living complex ecological system, acknowledging like many others that, though a definition might be precise and 'concrete', it is still technically very demanding to calculate in practice (Bartkowski et al., 2015; Jørgensen et al., 2016). To add complexity, all the dimensions of biodiversity are tightly interconnected, affecting the state and functioning of the ecosystem as well as the ESS (Laurila-Pant et al., 2015). Ecosystems are complex functional units, encompassing not only the biotic and abiotic components of the environment (i.e., the biophysical environment), but their ecology as well (i.e., how living organisms interact with each other and with the surrounding environment). To offer a consistent theory about ecosystem function a recent ecological sub-discipline has developed – Systems ecology (Jørgensen et al., op. cit.), which builds on four pillars (1) hierarchy, (2) thermodynamics, (3) networks, and (4) biogeochemistry (Jørgensen, 2012). Because of such complexity, it is not straightforward to account for the role of biodiversity or for the impacts of its decline on ecosystem services in general (TEEB, 2010b; Jørgensen and Nielsen, 2013; Laurila-Pant et al., 2015).

So the question is how to identify and select relevant proxies of biodiversity that allow moving current knowledge, attaining at some of the AQUACROSS main aims:

▸ Increasing our understanding of biodiversity and ecosystem functioning relationships (BEF);

▸ Understanding if BEF relationships patterns are common across all aquatic systems, and comparable to those identified for terrestrial ecosystems;

▸ Establishing causal links between biodiversity, ecosystem functioning and the provision of ESS;

▸ And applying an EBM approach to evaluate the impacts of anthropogenic activities in biodiversity, and ultimately in the provisioning of ESS, using the AQUACROSS AF.

There is still not a clear understanding of the underlying role biodiversity plays in ecosystem service provision (Kremen, 2005; Hattam et al., 2015). In order to understand this role, the parts of the ecosystem that provide the services need to be identified. Most studies consider parts of the ecosystem such as biotic groups (e.g., Grabowski et al., 2012), habitats (e.g., Burkhard et al., 2012) or functions (e.g., Lavery et al., 2013) in understanding the effect changes in these have on the supply of ESS. Interactions between multiple biotic groups or habitats (thus overall biodiversity) can influence service supply (Barbier et al., 2011). However, even where biodiversity generally has been related to the supply of services, this has started with identifying the initial relationship between specific biotic groups and their

supply of services and then considering biodiversity of these groups at a regional scale (Worm et al., 2006).

The task of understanding how biodiversity provides ESS is aided by a clear system to categorise and link the services derived from ecosystem components and the services and benefits provided by these components to the social system (the ecosystem service flow in Figure 11). This categorisation system helps to identify all the parts of the ecosystem that contribute to supply i.e. the important ESS providers (Kremen, 2005), through the flow of processes and functions leading to services and benefits. Once the ecosystem components that supply services have been identified, an understanding of the types of interactions between changes in ecosystem state and service generation is needed to assess the ecosystems capacity for service supply.Assessing biodiversity and evaluating the state of ecosystems requires suitable indicators for tracking progress towards environmental goals, for quantifying the relation between biodiversity and the function, and for establishing links with ecosystem services provision (e.g., Pereira et al., 2013; Tittensor et al., 2014; Geijzendorffer et al., 2015; Teixeira et al., 2016). But for assessments to contribute to increasing our understanding of the general causal links between BD–EF–ESS, it is crucial to ensure comparability of the biodiversity measures adopted (Pereira et al., 2013; Gonçalves et al., 2015; GOOS, 2016), by selecting at least a minimum set of common metrics for monitoring trends in biodiversity and the integrity of the ecosystems.

In the process of selecting operational indicators it is nevertheless important to emphasize what Jost (2006) so clearly stated: "a diversity index is not necessarily itself a "diversity"', and likewise the many measures used as proxies to grasp biodiversity, by themselves, are not biodiversity". This points to the need of using complementary measures that account for the complexity and many facets of biodiversity (Kremen, 2005; Borja et al., 2014; Bartkowski et al., 2015).

In Section 2.5.3 several potential sources of indicators (and indices or associated metrics) are presented. It is however important to have present that the field of biodiversity valuation is rather heterogeneous regarding both valuation objects and valuation methods (Bartkowski et al., 2015; Teixeira et al., 2016). The conservation and environmental management programmes have had different goals and approaches through time and have therefore selected different components to be assessed (see Section 2.2), leading to different classifications and to the choice of different indicators. For example, earlier conservation initiatives (e.g EU Nature and Water Directives) have focused traditionally on structural components individually, or in communities' composition and associations and habitats, which is then reflected in the classifications adopted (such as the EUNIS biotopes classification, species red lists, biological quality elements etc.). More recent EBM approaches (e.g., MSFD, EU Biodiversity Strategy) attempted to integrate the interplay between natural, social and economic systems, with their choice of indicators reflecting these different dimensions and the interactions between them (e.g., biodiversity, food webs, commercial fish and shellfish, contaminants, improved knowledge of ecosystems and their services). Such inconsistency between existing approaches leads to a gap in standardized classifications for

identifying the different and most relevant components of biodiversity for selecting biodiversity indicators, as is discussed in Section 2.5.3.

It is important to understand the parts of the ecosystem that deliver services (e.g., species, groups of species or habitats), and indicators should be reflective of these ecosystem components and their functions (Kremen, 2005; Hattam et al., 2015). One approach for doing this could be through starting with a typology of ecosystem components representing the ecosystem (e.g., habitats and their biotic groups) and using relative contributions to identify how much these components contribute and which are the most important ecosystem service providers (Burkhard et al., 2012; Robinson et al., 2014; Culhane et al., 2016; Sousa et al., 2016; Tempera et al., 2016). This approach can be qualitative, simply indicating whether a component gives a low, medium or high contribution (e.g., Burkhard et al., 2012; Robinson et al., 2014) or can be more quantitative, using information such as rates of relevant functions and spatial extents of the service providers (Culhane et al., 2016; Sousa et al., 2016; Tempera et al., 2016). Once the ecosystem service providers have been identified, these can be the focus for identifying indicators of the functions, benefits and services, while maintaining a strong link with the state of the ecosystem (as discussed in the following paragraphs for EF and ESS). A typology of ecosystem components can facilitate assessment of changes in state due to drivers and pressures and consequent changes in the supply of services by linking them to a typology of drivers and pressures upstream, (presented in Section 2.4) and to typologies of ESS downstream. An example can be found below in Tables 8 to 10 based on the Common International Classification of Ecosystem services (CICES) typology, and includes the ESS and the abiotic outputs from the ecosystem (links between ecosystem components and a marine specific CICES typology has been carried out in Culhane et al. (2016).

Ecosystem Functions

Any application of ecological models, selection of indicators, and quantifications of ESS requires a sound knowledge of how ecosystems are working as systems (Jørgensen et al., 2016), i.e. functioning.

However, the definition of ecosystem functioning and in particular the indicators used for measuring ecosystem function do not gather more consensus (Jax, 2005; Nunes-Neto et al., 2014; Dussault and Bouchard, 2016) than that found for biodiversity. The term "function" has been used in different ways within environmental science (Jax, 2005), and in particular within ecology (Dussault and Bouchard, 2016) and ESS context (Jax, 2016).

In ecology, functions have privileged a contextual and relational aspect, i.e. "causal role" functions (see discussion by Dussault and Bouchard, 2016), over an evolutionary perspective. Based on the organizational theory of functions, function in ecology has been defined by Nunes-Neto et al. (2014) as "a precise effect of a given constraint on the ecosystem flow of matter and energy performed by a given item of biodiversity, within a closure of constraints". This definition clearly distinguishes and links the different components of the biodiversity and ecosystem function (BEF). And in fact, in a EBM context, as that of the AQUACROSS AF, attributing functions to biotic and abiotic components of ecosystems facilitates the purpose

of analysing processes of an ecosystem in terms of the causal contributions of its parts to some activity of an ecosystem (Jax, 2005), for example related with ESS. Nevertheless, this approach may reveal itself to be insufficient with respect to some important aspects of the BEF research, namely in the relationship between biodiversity and ecosystem stability and resilience (Loreau and de Mazancourt, 2013; see discussion by Dussault and Bouchard, 2016). From an evolutionary perspective ecological functions should be defined relative to an ecosystem's more general ability to persist (i.e., both resistance and resilience). Accounting for how species traits enhance their present fitness and therefore their propensity to survive and reproduce (Bigelow and Pargetter, 1987) might better suit the focus of BEF research on the relationship between biodiversity and ecosystem resilience and sustainability. This, in turn, when scaled-up to ecosystems level, can be interpreted as a propensity to persist (i.e., in terms of ecosystem stability and resilience (Bouchard 2013a, 2014 in Dussault and Bouchard, 2016)).

In the context of AQUACROSS, ecosystem function is defined as "a precise effect of a given constraint on the ecosystem flow of matter and energy performed by a given item of biodiversity, within a closure of constraints. Ecosystem functions include decomposition, production, nutrient cycling, and fluxes of nutrients and energy". Ecosystem functions differ from ecosystem processes, as these are a "physical, chemical or biological action or event that link organisms and their environment. Ecosystem processes include, among others, bioturbation, photosynthesis, nitrification, nitrogen fixation, respiration, productivity, vegetation succession".

In the process of implementing an EBM approach, it is essential that the measures of ecosystem functioning can be correlated both with measures of biodiversity of ecosystems (Hooper et al., 2005; Cardinale et al., 2006) on one side and with measures of ESS (Harrison et al., 2014) on the other side. Section 2.5.3 will present the approach to the selection of EF potential measures to be adopted.

Ecosystem Services

In the scope of AQUACROSS AF, ESS are the final outputs from ecosystems that are directly consumed, used (actively or passively) or enjoyed by people (see Tables 8 to 10). In the context of CICES they are biologically mediated. (Human environmental interactions are not always ESS, e.g., maritime traffic, tourism activities). This concept tries to bring together previous definitions.

ESS have been defined as 'the benefits people obtain from ecosystems' (MA, 2005). The ecosystem service approach aims to recognise and make visible the value of nature (TEEB, 2010b), considering the 'direct and indirect contributions of ecosystems to human well-being'. In this context the concept of ecosystem goods and services' is synonymous with ESS. In the context of Action 5 of the EU2020 Biodiversity Strategy (Maes et al., 2014) the two previous definitions are acknowledged, and the service flow (see Figure 11) refers to the 'actually used service'. The benefits to society include those that are passively obtained but essential to human survival (e.g., climate regulation), those that are actively obtained and of

critical importance (e.g., food), and those that are not essential but enhance our existence either actively or passively (e.g., aesthetics) (Kremen, 2005).

While some assessments consider ESS from the supply-side, considering the capacity of the ecosystem to supply services (e.g., services supplied by kelp forests, Smale et al., 2013), others take an economic perspective, the demand-side (e.g., the value of recreational fishing, Toivonen et al., 2004), whilst some evaluate both supply and demand (e.g., Burkhard et al., 2012).

As part of the development of an operational AF, an operational definition of what an ecosystem service is, and how this relates to the ecosystem is required. The definition and typology of ESS has been identified as being an important criterion, but often a major weakness, in current frameworks for the assessment of ESS (Nahlik et al., 2012). In different studies, 'services' can sometimes refer to ecosystem functions, services or benefits (Böhnke-Henrichs et al., 2013). In the AQUACROSS approach, though, each of the steps will be clearly defined and kept separately (Figures 10 and 11), as also recommended by several authors (e.g., Potschin and Haines-Young, 2011; Liquete et al., 2013; Böhnke-Henrichs et al., 2013; Culhane et al., 2016; Lillebø et al., 2016).

In some cases the concept of ESS is considered as one part of the concept of 'natural capital', which is taken to include ESS, non-renewable resources and renewable resources (de Groot et al., 2010; EEA, 2015b). This implies, although not always explicitly stated, that ESS are the biologically mediated benefits that people get from nature i.e. that the service is underpinned by biological components and biologically mediated processes or functions. For example, in the application of ecosystem service typologies and assessments such as the Millennium Ecosystem Assessment (MA, 2005), The Economics of Ecosystems and Biodiversity (TEEB, 2010b) and System of Economic and Environmental Accounts (SEEA, 2012), in practice, these systems focus on those services that are underpinned by a connection to biodiversity and the biological processes and functions of the system (Haines-Young and Potschin, 2012).

This distinction between biologically-mediated services and abiotic outputs was recognised in CICES (Haines-Young and Potschin, 2012), which is the EU reference classification. While the role of biodiversity contributing to human welfare is fundamental, abiotic outputs (such as wind energy or minerals) also contribute benefits, and these should be accounted for (Haines-Young and Potschin, 2012; Lillebø et al., 2016). The benefits people get from ecosystems rely, to different degrees, on biological or abiotic parts of the ecosystem. In moving forward, CICES created a separate but complementary typology of abiotic outputs to facilitate their assessment, but in keeping with previous work (MA and TEEB), focused mainly on the biologically mediated services (Haines-Young and Potschin, 2012). While initially CICES recommended that all outputs are considered 'ESS' with a qualification specifying the level of dependency on biodiversity (Haines-Young and Potschin, 2012), the final iteration of CICES recommended that only those outputs reliant on living processes should be included as ESS, therefore excluding abiotic outputs from being considered ESS (Haines-Young and Potschin, 2012). This focus on biologically mediated services has been further emphasised through the adoption of the CICES classification system in the Mapping and Assessment of Ecosystem

Services (MAES) in Europe, which, so far, only considers the biologically mediated services for support of the EU Biodiversity Strategy i.e. those services which are associated with and dependent on biodiversity (Maes et al. 2013, 2014, 2016).

Despite this broad consensus in the current policy relevant assessments of ESS, it is recognised that this definition of services (biologically mediated) will not satisfy all and that future assessments would benefit from being integrated, accounting for biological and abiotic outputs of ecosystems (Haines-Young and Potschin, 2012). There are important arguments supporting the inclusion of abiotic outputs of the ecosystem, as they can have implications for spatial planning, management and decision-making (Armstrong et al., 2012; Kandziora et al., 2013; Sousa et al., 2016; Lillebø et al., 2016). The provisioning of services should reflect changes to ecosystem state (e.g., Böhnke-Henrichs et al., 2013; Haines-Young and Potschin, 2012). This means that to be considered a service, a change in state of the ecosystem can result in a change in the supply of a service. This is true of biologically mediated services, for example, a change in abundance of commercial fish populations has an impact on the supply of seafood. However, a change or a difference in the abiotic conditions can also lead to a change in the supply of abiotic services.

The AQUACROSS definition of ESS encompasses the goods and services people get from the ecosystem more broadly, such as the abiotic outputs that are not affected by changes in ecosystem state (e.g., oil and gas, salt, aggregates) (EEA, 2015b). The exploitation of abiotic outputs, in addition to the use of the ecosystem for economic activities (i.e., space for activities to occur), can have an impact on the state of the ecosystem and thus the potential supply of services, but are not affected themselves by the state of the biological components of the ecosystem. However, to build realistic scenarios for conservation and management purposes considering economic drivers, it is necessary to account for all services, namely the biologically mediated ESS and the abiotic outputs. In AQUACROSS, we aim to create a wide assessment of all services and benefits people get from nature, thus we include both the services dependent on biodiversity as well as those reliant on purely physical aspects of the ecosystem. The AF to be developed and tested within the AQUACROSS will account for both as in Tables 8 to 10.

Table 8: Ecosystem services, considering both biotic and abiotic dimensions, for the Provisioning category following CICES classification

Ecosystem services		Abiotic outputs from ecosystems	
Provisioning		Abiotic Provisioning	
Division	Group (includes the respective classes)	Group	Division
Nutritional	**Biomass** Wild plants and fauna; plants and animals from in situ aquaculture	**Mineral** Marine salt	Nutritional abiotic substances
	Water Surface or groundwater for drinking purposes	**Non-mineral** Sunlight	
Materials	**Biomass** Fibers and other materials from all biota for direct use or processing; genetic materials (DNA) from all biota	**Metallic** Poly-metallic nodules; Cobalt-Rich crusts, Polymetallic massive sulphides	Abiotic materials
	Water Surface or groundwater for non-drinking purposes	**Non-metallic** Sand/gravel	
Energy	Biomass	**Renewable abiotic energy sources** Wind and wave energy	Energy
		Non-renewable abiotic energy sources Oil and gas	

Source: adapted from Haines-Young and Potschin, 2012

Table 9: Ecosystem services, considering both biotic and abiotic dimensions, for the Regulating and maintenance category following CICES classification

Ecosystem services		Abiotic outputs from ecosystems	
Regulating and maintenance		Regulating and maintenance by abiotic structures	
Division	Group (includes the respective classes)	Group	Division
Mediation of waste, toxics and other nuisances	Mediation by biota	By natural chemical and physical processes	Mediation of waste, toxics and other nuisances
	Mediation by ecosystems Combination of biotic and abiotic factors	Atmospheric dispersion and dilution; adsorption and sequestration of waters in sediments; screening by natural physical structures	
Mediation of flows	Mass flows	By solid (mass), liquid and gaseous (air) flows	Mediation of flows by natural abiotic structures
	Liquid flows	Protection by sand and mud flats; topographic control by dunes and cliffs of wind erosion	Mediation of flows by natural abiotic structures
	Gaseous/air flows		
Maintenance of physical, chemical, biological conditions	Lifecycle maintenance, habitat and gene pool protection	By natural chemical and physical processes Sea breezes	Maintenance of physical, chemical, abiotic conditions
	Pest control		
	Soil formation and composition		
	Water conditions		
	Atmospheric composition and climate regulation		

Source: adapted from Haines-Young and Potschin, 2012

Table 10: Ecosystem services, considering both biotic and abiotic dimensions, for the Cultural category following CICES classification

Ecosystem services			Abiotic outputs from ecosystems	
Cultural			Cultural settings dependent on aquatic abiotic structures	
Division	Group (includes the respective classes)		Group	Division
Physical and intellectual interactions with biota, ecosystems, and seascapes [environmental settings]	**Physical and experiential interactions**			Physical and intellectual interactions with land-/seascapes [physical settings]
	Experiential use of biota and seascapes; physical use of seascapes in different environmental settings	Experiential use of seascapes; physical use of seascapes in different physical settings		
	By physical and experiential interactions or intellectual and representational interactions			
	Intellectual and representational interactions Scientific; education, heritage; aesthetic; entertainment			
Spiritual, symbolic and other interactions with biota, ecosystems, and seascapes [environmental settings]	**Spiritual and/or emblematic** Symbolic; sacred and/or religious			Spiritual, symbolic and other interactions with land-/seascapes [physical settings]
	Other cultural outputs Existence; bequest			

Source: adapted from Haines-Young and Potschin, 2012

As above, the ecosystem service approach aims to recognise and make visible the value of nature (TEEB, 2010b), considering the 'direct and indirect contributions of ecosystems to human well-being'. However, an increasing number of authors (e.g., Mace et al., 2011; Haines-Young and Potschin, 2012; Potts et al., 2014) have followed Fisher et al. (2008) who nest within the broad definition of ESS 'final' services and 'intermediate' services. In the context of the EU Biodiversity Strategy to 2020 the flow of ESS refers to the 'actually used service', the 'final' services. The rationale for this division is to avoid the double counting of intermediate (or supporting) services in the valuation step of the process.

The CICES classification of services (Haines-Young and Potschin, 2012) provides the following definitions for 'final services':

"Final ecosystem services are the contributions that ecosystems make to human well-being. These services are final in that they are the outputs of ecosystems (whether natural, semi-natural or highly modified) that most directly affect the well-being of people. A fundamental characteristic is that they retain a connection to the underlying ecosystem functions, processes and structures that generate them".

Despite some typologies explicitly stating that the typology consists only of final services, there is still ambiguity related to some the actual services included, in particular regulation and maintenance services, as there are different interpretations of services and whether they are final or intermediate. These different interpretations may be required for different assessments in different contexts (Hattam et al., 2015). This debate has not reached consensus in the literature and is on-going. However, we do not consider this to be prohibitive in the development of an operational framework which includes ecosystem service assessment provided there is the awareness of the potential for any double counting at the valuation stage. Furthermore, the CICES classification, which arguably includes intermediate services, is nevertheless comprehensive and, thus, a better reflection of all of the ways the ecosystem benefits society than other typologies, which are approached strictly from an economic-valuation perspective.

In summary, hardly any artificial "classification will be able to capture the myriad of ways in which ecosystems support human life and contribute to human well-being" and "no fundamental categories or completely unambiguous definitions exist for such complex systems" (de Groot et al., 2010).

Supply versus demand of ecosystem services

The assessment of ESS can be approached from the supply side – the potential or capacity of the ecosystem to supply services, whether or not it is used, or the demand side – the services people ask from the ecosystems whether they are actually provided or not (see Section 2.1). One can say, therefore, that a 'supply side' assessment based on ecosystem capacity considers how the state of the ecosystem is affecting the generation of the actually used services (Burkhard et al., 2012) and the potential to provide more and better services for present and future generations.

While the capacity of the ecosystem to supply services is tightly linked to the state of the ecosystem (biodiversity and ecosystem processes and functions), the demand and actual use of services can be decoupled from the state of the ecosystem, as they are a clear outcome of social processes. For example, a study of recreational clam digging found most activity occurred at easily accessible sites (where there were parking facilities) even though more valuable stocks were present in other (less accessible) locations (O'Higgins et al., 2010).

A change in ecosystem state and biodiversity can lead to a change in the supply of services but not in the demand of services. However, the detrimental impacts of the use of services can, in turn, lead to a change in ecosystem state and biodiversity and to a change in the supply of services. The demand for ESS, including the use of abiotic natural capital (e.g., aggregates), or the use of ecosystem 'space' for economic activities – can affect the supply of

ESS through alterations in the state of species and habitats and biodiversity overall, and are considered in Section 2.4.

2.5.3 Identification of relevant indicators and associated metrics

As part of the development of an EBM operational AF, classification methods to be applied to each compartment (i.e., BD, EF, and ESS) are required, which enable furthermore establishing links between each other. As well, a clear and common understanding of the concepts of indicators, indices and metrics is required.

Since ESS depend on the ecosystem functions provided by biodiversity, there is a need for ecosystem-based approaches consider the causal links between biodiversity, ecosystem functioning and ESS, and also a need to identify indicators and metrics relevant for aquatic ecosystems that may be used to establish their state. A clear definition of each part of these BD-EF-ESS relationships facilitates therefore the identification of appropriate indicators (Böhnke-Henrichs et al., 2013; Liquete et al., 2013). One of the advantages of having a set of indicators is that they aid organizing the type of information needed for the assessment, and also allow quantifying the relationships between the different components and the flows across the AF. Indicators can also provide insight into variations in resilience by reporting e.g. on ecosystem recovery rates after disturbance (Lambert et al., 2014; Rossberg et al., 2017). This in turn can be used to assess the sustainability of human activities' impacts and support the development of appropriate management strategies (Lambert et al., 2014; Lillebø et al., 2016).

However, even with clearly defined and segregated components (Böhnke-Henrichs et al., 2013; Liquete et al., 2013) of the BD-EF-ESS (Section 2.5.2), the complexity of the ecological systems, where structure and processes combine in a myriad of ways to perform functions and to secure ESS supply, still makes the selection of indicators a difficult process in practice (e.g., Maes et al., 2014; Lillebø et al., 2016).

Guidance will be provided for selecting biodiversity components, ecological functions and ESS and respective indicators in ways that the assessment reflects the complexity of social-ecological interactions (Gómez et al., 2016; Saunders and Luck, 2016). It is therefore crucial that the processes described in Section 2.4 are also accounted for in order to achieve a meaningful selection of ecosystem components and associated indicators. In addition, having a list of indicators, as comprehensive it may be, does not ensure by itself a coherent evaluation of how the ecosystem state and functioning converge to secure the supply of ESS. Nevertheless, criteria to identify and test the quality of indicators are available and can be used (Heink et al., 2016), namely the framework from the Biodiversity Indicators Partnership[22]; and the framework to test quality of indicators proposed by Queirós et al. (2016).

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[22] For more information visit: http://www.bipindicators.net/

This guidance aims also at promoting consistency throughout the case studies, such that a standardized approach may ultimately allow a comparison of BEF and BES relations identified across aquatic realms, contributing to understand whether they are interchangeable or ecosystem-specific (see Section 2.5.4). To operationalise this, the guidance will focus on:

a) Defining comprehensive classifications (and developing relevant subcategories) pertinent for aquatic ecosystems, within each main theme: i.e. Biodiversity, Ecosystem Functions, and Ecosystem Services, since such subcategories will allow building meaningful causal networks between the different components of the framework. The classification systems will be tailored to the AQUACROSS needs, either by building on scattered approaches (as for Biodiversity and Ecosystem State assessment), or by developing new ones (as in the case of Ecosystem Functions), or by adapting existing ones (as the CICES Ecosystem Services classification enlarged to accommodate abiotic outputs).

b) Providing lists of indicators, and/or sources of indicators, and allocate indicators within each theme classification (i.e., BD, EF and ESS) and respective subcategories;

c) Identifying criteria for the selection of good indicators, relevant within each theme, and setting a *de minimum* approach to be applied across case studies;

d) Providing recommendations for applying a holistic approach to the BD-EF-ESS, accounting for interactions, synergies, and trade-offs, when identifying causal links.

Comprehensive classifications will be proposed, or adapted, for:

▸ Biodiversity: the ecosystem components to be considered will build on the requirements of the various environmental policies in place, but will allow to adjust to case studies needs, accounting also for scale issues;

▸ Ecosystem Functions: the most relevant functions in aquatic ecosystems will be identified, together with the associated ecological processes. While in our proposal the Ecosystem Functions categories defined are distinct and exclusive, the underlying ecological processes might be linked to more than one function category. A subsequent selection of appropriate EF indicators will measure the contribution of the function to providing an ecosystem service.

▸ Ecosystem Services: AQUACROSS will follow the recently adopted MAES typologies of ESS (Maes et al., 2013), which build on latest version (V4.3) of the CICES approach (Haines-Young and Potschin, 2012; Maes et al., 2014, 2016), as this ensures comparability with the approaches being followed by Member States. CICES differs from the previous ESS classifications, namely MA and TEEB, in that, to avoid double counting, it recognises only three categories (called 'sections') of 'final' ESS: provisioning services, regulation and maintenance services, and cultural services. In addition, we propose that the CICES accompanying matrix of abiotic outputs from the ecosystems are also taken into consideration when applying the AQUACROSS AF. In this sense, listed categories are extended as shown in Tables 8 to 10.

Indicators

Preliminary lists of indicators (and associated metrics, accompanied with the respective definitions) have been elaborated accounting for indicators outlined by key legislation identified in the project and identified in relevant scientific literature. For each component of the BD–EF–ESS relationship the possible sources and examples of indicators will be referred. However these are not intended to be prescriptive lists and each case study should select the indicators deemed more adequate for the context and purpose of study (i.e., the aquatic realm, the ecosystem features, the scale(s) of study, the identified pressure(s), the ESS being scrutinized). This means that the selection of indicators at this stage should be integrated and in line with the other stages of the AF, so that a successful flow of information is achieved (see Section 2.1).

As discussed in Section 2.4, indicators for the DPS (state in the AQUACROSS AF encompasses BD and EF) part of the DPSIR will be established based on the method developed for marine ecosystems in the ODEMM project (see Section 2.4 for more detail). Subsequently, it will be ensured that the indicator list developed under this stage of the assessment (i.e., for biodiversity, ecosystem functioning and ecosystem services) is compatible and can be linked to the DPS components of the previous stage. However, as discussed in 2.5.1, state metrics will not always align between appropriate metrics to assess the change in state due to a pressure, and those metrics appropriate for assessing the ecosystem's capacity to supply services.

Regarding biodiversity and ecosystem functioning, numerous indicators and indices are available for assessing the state of aquatic ecosystems (see for example the following reviews: Piet and Jennings, 2005; Piet et al., 2006; Birk et al., 2012; ICES, 2014, 2015; Hummel et al., 2015; Piroddi et al., 2015; Teixeira et al., 2016), often developed in response to legal requirements (e.g., the Water Framework Directive (WFD), the Marine Strategy Framework Directive (MSFD), the EU2020 Biodiversity Strategy SEBI indicators, the Red List Index for European species and the Habitat Directive (HD)). Thus, for aquatic ecosystems, it will be essentially based on the requirements set by these legal frameworks that Member States will map and assess the state of their ecosystems, as required also by the EU Biodiversity Strategy 2020 Action 5. The adoption of such indicators within the case studies when applying the AQUACROSS AF not only favours a relevant link with European policy, but ensures also that data are likely to be available for indicators and metrics referenced within those legal documents (Hummel et al., 2015; Berg et al., 2015; Patrício et al., 2016a and b; Teixeira et al., 2016).

Available indicators include those from structural to functional approaches, ranging from the sub-individual level to the ecosystem level, and capturing changes and processes operating at different spatial scales. The scope of the indicators available is thus wide and therefore it should be able to cover the needs of the different case studies' needs. Nevertheless the development of new indicators development could be justified within the AQUACROSS project, and would complement gaps in the existing resources. This might be particularly relevant in the case of functional indicators, traditionally not incorporated in applied

management, but where recent research is thriving with new approaches to measure functionality (see Section 2.5.2).

Regarding ESS, an initial list of indicators was obtained from the comprehensive review elaborated by Egoh et al. (2012) and complemented with the recent list of MAES indicators for ESS (Maes et al., 2014, 2016), and with Hattam et al. (2015) specific indicators for marine environment. Also, to accommodate the inclusion of abiotic outputs, potential indicators will be identified and added to the lists. As mentioned before, the selection of specific ESS indicators will be driven by the case studies' context and needs.

Lessons learnt from this application of indicators to the showcase case studies may lead to an adaptation of the AF and/or the overall concepts of AQUACROSS.

2.5.4 Methods to analyse causal links

Among the multitude of available multivariate analysis tools and methods, discriminant analysis (DA) may be used to examine relationships between both nominal and continuous variables. Like many other multivariate methods, DA tries to reduce statistical dimensionality by extracting the dominant gradients of variation from a set of multivariate observations. However, the most distinctive aspect of DA is that it allows a priori designation of samples into groups. DA weights the contribution of variables by their effectiveness in minimizing the difference within each predefined group while maximizing differences among groups (e.g., Palmer et al., 2009). DA will hence allow to optimally comparing data from existing BEF and BES studies available in the open literature by considering variables that influence reported BD–EF–ESS relationships (e.g., those identified from the literature review mentioned in Section 2.5.1).

In the past decade, several meta-analyses on data obtained from manipulative experimental BEF experiments have been conducted to attain evidence for BEF relationships (Cardinale et al., 2011). By considering the variables identified from the DA, a refined meta-analysis on data from existing BEF and BES experiments may be conducted. This may thus be expected to lead to less variable functions and hence a more precise estimation of the causal links between biodiversity, ecosystem functioning and ecosystem services.

The most suitable metrics and models from previous analyses (see also Section 2.5.3) will be selected for integration in the management tool by direct integration if relevant models are identified, or after informing Neural Networks Models to forecast results of biodiversity causality links. Causality functions linking biodiversity and ecosystem functions, derived with the help of these modelling frameworks, will be integrated into the ARIES ecosystem services modelling platform to characterise the link between ecological function and societal benefits and to provide a bridge to ESS trade-off analysis.

A qualitative assessment of uncertainty will be carried out involving:

▸ Compilation of a comprehensive list of all possible sources of uncertainty in each model (e.g., Mastrandrea et al., 2011);

▸ The use of classification categories to help produce a list (listing will be prioritized over classification);

▸ The introduction of other categories as necessary;

▸ Components of the full modelling chain will be used to enable uncertainty to be fed through.

The multidimensional nature of causality relationships will be addressed with multivariate modelling approaches to derive biodiversity and ecosystem functions and services across large regions. Several experimental approaches have been used in the last 20 years to demonstrate causality links between BD–EF–ESS. Since biodiversity has already been recognised as a multidimensional concept, BEF assessment will not rely solely on species richness but will also consider the functional trait composition of biological assemblages using multi-metric biodiversity indices (c.f. Sections 2.5.2 and 2.5.3). These will consider aspects related to species composition whose importance is demonstrated, e.g. i) relative contribution of dominant vs. minor species, ii) environmental context, iii) density dependence and community structure. AQUACROSS will also explicitly incorporate, to the extent possible, the causal effects of structuring abiotic (environmental heterogeneity) and biotic (movement, dispersal) processes that are key to species co-existence and vital to the maintenance of species diversity. The multidimensional nature of causality will be addressed with multivariate modelling approaches and used to derive BEF across large regions through the use of generalised dissimilarity modelling (GDM) and generalised diversity–interactions models (GDIM) as they are nonlinear models that address effects of species interactions on biodiversity patterns. GDM will be used to analyse spatial patterns of turnover in community composition (beta diversity), across larger regions, while accommodating the types of nonlinearity commonly encountered in large-scale ecological data sets. This will facilitate dealing with ecological gradients and associated ecosystem functions.

Integrated models will also be established as they can highlight priorities for the collection of new empirical data, identify gaps in our existing theories of how ecosystems work, help develop new concepts for how biodiversity composition and ecosystem function interact, and allow predicting BEF relations and its drivers at larger scales (Mokany et al., 2015; Queirós et al., 2015). Such models could also form components within larger 'integrated assessment models', improving consideration of feedbacks between natural and socioeconomic systems (Mokany et al., 2015), ultimately aimed at better informing management as is seen in the framework underlying the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) (Díaz et al., 2015).

2.6   Crosscutting issues

While previous sections of this document outline different elements of the AQUACROSS analytical approach, certain aspects of analysis may benefit from a common macroscopic overview. This Chapter provides an overview of commonalities in data structure under the AQUACROSS Information Platform (hereafter IP) as well as considering some common sources of uncertainty and presenting some conceptual tools that can be used qualitatively or quantitatively to facilitate the analysis of systems adopting a 'bigger picture' approach.

Besides dealing with uncertainty, some major challenges on how sources of datasets could affect understanding are shown in Syphard et al. (2011) and O'Higgins et al. (2014a and b) regarding different spatial and temporal resolution; Yesson et al. (2007), Jetz et al. (2012), Kwon et al. (2016) and La Salle et al. (2016) regarding data integration of various taxon-level data types (genome, morphology, distribution and species interactions) as well as spatial and temporal scales and Vandepitte et al. (2010, 2015) regarding quality control criteria.

2.6.1 Introduction: going beyond data and metrics – information flows for analytical purposes

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Making the AQUACROSS Concept operational requires the integration of information within the AQUACROSS Architecture (see above 3) at the different spatial and temporal scales. The information layers are a key component of the AQUACROSS Architecture for analysing the complex interaction between social and ecological systems and finding effective, efficient, and socially acceptable EBM responses (Gómez et al., 2016).

The AQUACROSS IP is the central entry point for project partners and scientists to publish and share the data on different types of aquatic ecosystems, biodiversity and EBM practices. A significant amount of the data that will be gathered through the different project case studies will have a spatial component. Hence, the AQUACROSS IP should be able to provide all the functionalities for managing spatial data in an efficient and flexible way. The ultimate goal of the AQUACROSS IP will be to share the scientific knowledge on Aquatic EBM by means of a Spatial Data Infrastructure for aquatic ESS and biodiversity connected to other existing relevant information platforms.

The AQUACROSS IP is based on the open-source CKAN data portal platform. CKAN is a tool for making open data websites. It contains a powerful data management system and is aimed at data publishers wishing to make their data and associated metadata open and available, helping to manage and publish their data. Once data is published, users can use its multi-faceted search features to browse and find the data they need, and preview it using maps, graphs and tables. National and local governments, research institutions, and other organisations that collect data use CKAN. CKAN is currently the technical solution

implemented by the European Commission to publish pan-European open datasets across the EU.

Within CKAN, data is published in units called "datasets". A dataset contains metadata about the data, and one or more "resources" which hold the actual data. CKAN can accept data in any format, including the formats of CSV or Excel spreadsheets, XML, PDF, images, RDF, GeoTIFF, Shapefiles, etc. CKAN can store the resource internally or store it simply as a link to external resource on the Internet. These CKAN resource page can contain one or more visualisations of the resource data or file contents (a table, a bar chart, a map, etc), which are commonly referred to as resource views.

AQUACROSS information

The AQUACROSS datasets are divided into thematic categories following the AQUACROSS AF:

1   Drivers of change and pressures on aquatic ecosystems

2   Biodiversity, ecosystem functions and ESS

3   Assessment of scenario and prioritisation measures

4   Ecosystem-based Management towards policy objectives

The following categories describe typical and potential datasets under each category. The flows and links with the categories follow the relational chain proposed in the AF, which aims to facilitate the identification and selection of the information and indicators. The chain begins with a high level driver, the economic activity (direct driver), and associated pressure and a part of the ecosystem where that pressure can cause a change in state. This structure allows the selection of indicators of ecosystem state or pressure and identifies the human activity, which can be a focus of management (see Sections 2.4 and 2.5).

Category 1: Data on drivers, pressures affecting aquatic ecosystems

Drivers of ecosystem change

This category will include the information related to the drivers that can change the structure and function of ecosystems and their capacity to provide ESS to meet the demand of food, energy, transport, space, tourism services and many other goods and services. The information used to analyse the drivers of ecosystem change are mainly based on the economic sectors or activities that benefit from the provision of water related ESS. This includes their value added (e.g., Gross Value Added (GVA) of fisheries, agriculture, value added of the maritime manufacturing sector); employment per economic activity (e.g., in agriculture, fisheries, aquaculture, coastal tourism) and the use of provisioning ESS (e.g., freshwater resources per inhabitant, groundwater and surface water abstraction, tourism accommodation establishments, volume of goods handled in maritime transport, extraction of salt, fish). All drivers of ecosystem change must be properly understood at different

temporal and spatial scales (Section 2.4). Data sources on drivers of ecosystem change cover many statistics produced by Eurostat, national and regional statistics offices on water use by economic sector, fisheries, aquaculture, catch, maritime transportation, labour productivity, consumption and investment and statistical classification of economic activities (NACE).

Pressures over ecosystems and biodiversity

According to the AQUACROSS Concept and the AF, significant pressures are those that result in a change in ecosystem state leading to a change in the functioning of the ecosystem and thus can impact both biodiversity and human welfare. Pressures can be physical, chemical or biological. The linkage framework integrated into the AQUACROSS Architecture (see above 3) facilitates the identification of indicators needed to describe the system from the demand side. The AQUACROSS AF requires the identification of the pressures for each driver and type of activity identified. The information on pressures across aquatic realms should cover the pressures categories proposed in the AF: biological disturbance, chemical change, hazardous substances, physical change, pollution and climate change. The information related to driver-pressure across the different aquatic realms would be essentially based on the reported data under the Water Framework Directive (WFD), Habitats Directive (HD) and the Marine Strategy Framework Directive (MSFD). For example, fishing mortality, occurrence and spatial distribution of invasive species, nutrients concentration in the water column are some of the indicators proposed in the Good Environmental Status (GES) to measure the pressures on the marine environment.

Status of the ecosystems and biodiversity

Data on the assessment of the biodiversity and ecosystem state are also sourced from EU reporting obligations: WFD, MSFD, HD. At European level, state indicators are reported in the context of the European Environmental Agency a core set of indicators, SEBI indicators, the WFD ecological status, environmental status (MSFD), species and habitat conservation status and red list index (HD art. 17). Under the marine ecosystem and biodiversity, the GES descriptors indicators are strongly related to biological quality elements that indicate the integrity of the ecological system. Examples of dataset are on the abundance and distribution of selected species, population abundance, habitat distribution and physical, hydrological and chemical condition.

For instance, data collection under WFD has outputs and challenges such as intercalibration of river basins, gaps for transitional waters, multi-pressure context, or taking into account uncertainties (Reyjol et al., 2014).

Beside the reported data by the Member States, there are several relevant sources that could be used for the assessment of pressures and ecosystem and biodiversity status. These data can be found on scientific data portals, such as, the European Marine Observation and Data Network (EMODnet), Copernicus Marine environment monitoring service and national and regional Spatial Data Infrastructures (SDI) (e.g., Marine Scotland, Welsh Government, REDIAM-

Andalucia). O'Higgins et al. (2016) have reviewed many of the relevant aspects of European Spatial Data infrastructure.

Project partners will be requested to post links to these data sources on the IP. The majority of these datasets are expected to be available through both OGC web services and available for download in raster or shapefile formats.

Category 2: Data on ecosystem functions and services

Ecosystem functions (EF)

This category will include the information related to the ecosystem functions of the aquatic ecosystems. Ecosystem functions are usually organised into three categories: 1) production; 2) biogeochemical cycles; and 3) structural. Ecosystem functions related information quantity the stocks of materials and rates of processes involving fluxes of energy and matter between trophic levels and the environment, for example, nutrient levels, water retention of soils, water and air purification, habitat provision, carbon sequestration, extension and health of seagrass, among others.

Novel issues, such as remote sensing characterization of ecosystem functioning, are likely to come up in further discussions about information and data (Cabello et al., 2012).

Ecosystem services (ESS)

This category will include information related to the ESS assessment. Under the category of the Ecosystem Services the information derived will follow the CICES (Haines-Young and Potschin, 2012), the indicators and metrics were categorized using the EU MAES ESS categories adopted from the CICES: 1. Provisioning; 2. Regulating and Maintenance; and 3. Cultural. The dataset will also include indicators for the current assessment of ESS, divided into:

▶ Freshwater: e. g. water consumption for drinking, freshwater aquaculture production, surface water drinking, water abstracted, nutrients loads, waste treatment, sediment retention, flood prevention, Carbon sequestration (riparian forest), among other indicators;

▶ Coastal and Marine: e.g. nutrient load to coast, heavy metal and persistent organic pollutants deposition, oxygen depletion risk, composite indices based on extent of selected emerged, submerged and intertidal habitats, coastline slope and coastal geomorphology, wave regime, tidal range, relative sea level, storm surge, species distribution, C stock, C sequestration, presence of iconic/endangered species.

Besides this differentiation by realms, in order to integrate them, it may be meaningful to consider the inclusion of datasets, maps and indicators that refer to the delivery of ESS – supply side, flows of ESS– and to the demand side – benefits of ESS–, following the AQUACROSS Architecture.

However, since ESS delivery is functionally interlinked, there are difficulties on mapping ESS provision linked to individual ecosystem functions (e.g., Rees et al., 2012). Remote sensing data collection and use could be helpful to produce spatially-explicit assessments and valuation of ESS, (e.g., Araujo et al., 2015).

Finally, the differing data priorities between scientists and decision-makers could likely result in dataset biases or affect IP usefulness (e.g., Goldsmith et al., 2015).

Category 3: Related data on the assessment of scenario and prioritisation measures

This category will include the main outcomes of the assessment of scenarios including the results from modelling aquatic biodiversity and ESS to forecasting their development based on different scenarios and /or the optimisation of their spatial allocation in the case study. The data derived under this category depends very much on the case study and the development of scenarios, i.e predictive scenarios with limited choices of relevant driver variation for currently existing models or explorative and normative scenarios integrating social-ecological dynamics. The datasets we envisage under this category are largely the ones that will be generated in the course of the AQUACROSS project. Nevertheless, we mostly expect the outcomes to be available as map products. The information derived might refer to changes in drivers (e.g., increase of the domestic water abstraction according to population projections) or to the implementation of alternative policies (such as setting biodiversity strategy targets). This includes, for example, data on projected drivers and pressures, the potential habitat suitability of a species in a given area; data on their impact on the ecosystem functions and services, on different priority areas according to actions needed to overcome biodiversity impacts (or protect current biodiversity) and the actions needed to provide the targeted ESS delivery.

Some illustrations of examples above are included in the literature (e.g., Bocedi et al., 2014; Candela et al., 2016; Pistocchi et al., 2016). Additionally, the combination of data and models for cross-scale comparisons might be considered (e.g., Legendre and Niquil, 2013; Thuiller et al., 2015).

Category 4: Ecosystem-based management towards policy objectives

This dataset will include the main outcomes of the EBM to the case studies, including all types of information and indicators that will help monitoring and informing policy-makers about the effects of the responses or actions taken by society, individually or collectively. More specifically, these indicators will provide information on designated areas for policy intervention, such as multi-zoning planning according to the spatial prioritization of the different scenarios. In addition to this, distance-to-targets indicators could also be used to quantify the efforts required to reach policy targets and compare the differences of the scenarios in terms of important areas of biodiversity and the provision of ESS.

Examples on how datasets and information can help EBM and decision-making, are shown in Edwards et al. (2010), Hawkins et al. (2013), or Clavero and Villero (2014), regarding time-series, and in Fontaine et al. (2015), regarding taxonomic resolution in datasets when delineating conservation units.

While the AQUACROSS IP provides a location for the categorisation of data according to the themes described above as well as incorporating geographic information, it cannot provide the analysis required to integrate this information. For example, the EBM datasets (category 4) will be developed based on analyses run outside of, and subsequently uploaded to, the platform. There are a number of papers dealing with different analyses to apply to datasets (e.g., Boehme et al., 2014).

Regardless of the quality and availability of the case study data and the modelling tools for the project; uncertainties are unavoidable throughout the analytical process.

2.6.2 Dealing with uncertainty

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Uncertainty is a critical factor at different stages of the assessment process. This section is intended to provide analytical approaches to address uncertainty and achieve robust solutions.

Scientific support of societal decisions consists in investigating and communicating the degree of fulfilment of societal objectives achieved by suggested management alternatives (including the alternative of not taking an "explicit measure" and continuing with the past policy). This includes the creative process of finding alternatives that may have the potential of a high degree of achievement of the objectives. The process of scientific decision support is affected by many sources of uncertainty and it is important to consider and communicate these uncertainties in the decision support process.

There are three main sources of uncertainty in societal decision support or policy advice:

1   *Uncertainty about societal preferences*.
    Societal preferences can be derived from a policy analysis or can be elicited from the public or from representative stakeholders. Both processes bear uncertainty. Analysing policies might include some scope of interpretation, different policies might contradict each other, and it might be unclear how to trade-off between them.

    Since the society consists of a large number of individuals with diverging interests it is challenging to quantify and integrate their preferences.

2   *Uncertainty about the effect of suggested management alternatives*.

    The estimation of the effect of management alternatives needs knowledge about the future socio-economic development, about the resulting changes of factors influencing

the system under investigation directly, and about the response of ecological and economic attributes to the alternative for given external changes.

3 *Uncertainty about the implementation of the chosen management alternative*.

Implementation of some alternatives may be more difficult than others for political or technical reasons. Ideally, this should be assessed also.

These types of uncertainty occur related to different components of the SES. Figure 12 illustrates the location of the three main sources of uncertainty along the DPSWR causal chain.

In this section, we suggest how to consider these three main sources of uncertainty, we then discuss the evaluation of alternatives given these uncertainties, and we conclude with a "checklist" of how to deal with uncertainty in AQUACROSS. Most of the material summarized in this section is based on a recent review by Reichert et al. (2015) and the literature cited therein.

Uncertainty in societal preferences

In contrast to scientific prediction, which we try to make as objective as possible, societal preferences are subjective by nature and also change over time. In the current context, we mostly assume to support decisions within time frames over which the societal preferences do not change significantly. However, when suggesting adaptive management to consider the acquisition of scientific knowledge when revising earlier decisions, changed societal preferences can be considered at the same time.

As outlined in Reichert et al. (2015), there are strong arguments for formulating individual or societal preferences as value or utility functions from decision analysis (Eisenführ et al., 2010). Value functions quantify the degree of fulfilment of an objective on a scale from zero to unity. They are very flexible regarding the functional dependence of these preferences from attributes of the system under study. Utility functions in addition consider risk attitudes and can be built on elicited value functions (Dyer and Sarin, 1982; Reichert et al., op. cit.). The use of this description of preferences makes it possible to base decision support on Multi-Attribute Value Theory (MAVT) or, when extended to utility functions, on Multi-Attribute Utility Theory (MAUT).

Uncertainty ranges of elicited trade-offs used to specify value functions can either be elicited from stakeholders or estimated based on elicited value functions from different stakeholders that are merged to a "societal value function". The resulting ranking of alternatives according to decreasing values or expected utilities can then be analysed regarding its sensitivity to these uncertainty ranges of the value functions.

Figure 12: Location of source of uncertainly along the DPSWR causal chain
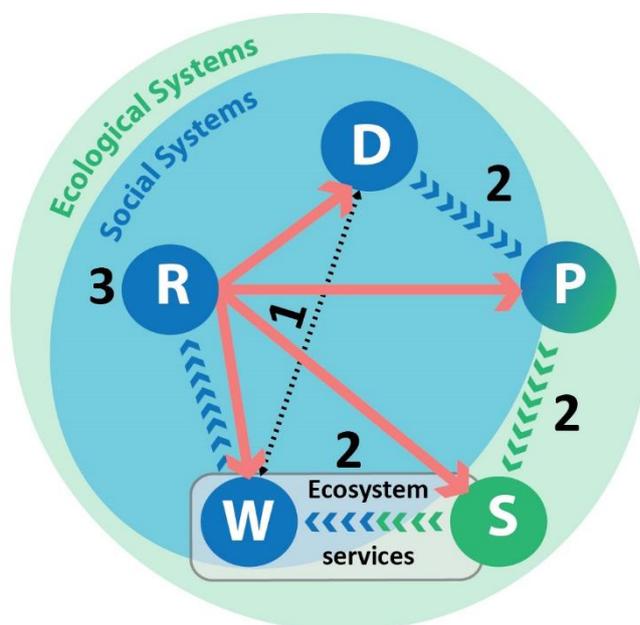
Source: O'Higgins et al., 2014a

Legend: 1– uncertainty about societal preferences. 2 Uncertainty about the effects of management alternatives. 3 uncertainties about the effectiveness of management.

Uncertainty about the effect of suggested management alternatives

In contrast to the societal preferences, the estimation of the consequences of suggested management alternatives and the uncertainty of these predictions is a scientific task that should be done as objectively as possible. Depending on the nature of the alternative (at the political, engineering or ecosystem manipulation level) this can involve social, engineering and natural sciences.

As outlined in Reichert et al. (2015), there are many arguments in favour of describing uncertain scientific knowledge through a probabilistic framework. One of the main reasons is that this framework can be used to describe random (due to non–deterministic behaviour of the systems) and epistemic (due to our lack of detailed knowledge about all relevant mechanisms in the system) uncertainty of the behaviour of a system in a compatible way. This is important as aleatory uncertainty becomes epistemic once the random event is realised but the outcome was not yet observed. In addition, the probabilistic framework easily allows us to formulate conditional probabilities, which is very important when considering future scenarios or policy alternatives (Cox, 1946). Finally, the argument of avoiding sure loss if probabilities are made operational with indifference between lotteries, adds another argument for using the probabilistic framework (Howson and Urbach, 1989).

To ensure the use of the best scientific knowledge, intersubjective probabilities should be used (Gillies, 1991, 2000; Reichert et al., 2015). This means that prior probabilities should be defendable by being supported by multiple experts or literature references. In case of very high ambiguity, sets of probability distributions, so-called imprecise probabilities (Walley, 1991; Rinderknecht et al., 2011, 2012, 2014), can be used instead of precise distributions. Bayesian statistics provides an ideal methodological framework for updating prior probabilities with actual observed data of a system.

There are three main contributions to uncertainty about the consequences of suggested alternatives:

a) Uncertainty about future socio-economic development.

b) Uncertainty in the prediction of future (environmental) influence factors that might change in response to the future socio-economic development.

c) Uncertainty in the response of ecological and economic attributes (indicators for the fulfilment of societal objectives) to management alternatives and the future environmental influence factors.

These three contributions to uncertainty are addressed as follows:

a. As the uncertainty about future socio-economic development is very large, it is best addressed by specifying potential scenarios for future development without specifying probabilities for these scenarios.

b. Prediction of the future behaviour of (environmental) influence factors can then be made conditional on these scenarios. In the current context, this will mainly consist of a compilation of existing information (expert opinions, results of published studies, published model results, etc.) in a probabilistic framework.

c. Prediction of the response of ecological and economic attributes relevant in the specific decision context will mainly be done by constructing models based on known mechanisms from the literature or from experts. Uncertainty is then considered through parameter uncertainty, intrinsic stochasticity of the model, and input uncertainty. Regarding the external influence factors, input uncertainty is given by the results from step b. Ideally, the model is formulated based on prior knowledge (often across similar systems) and updated based on observed data of the investigated system.

If observation error is large (which is often the case for ecological systems), it is advisable to explicitly distinguish model and observation uncertainty. Inference can then be done by using the model including observation uncertainty and prediction to describe our knowledge about the true state rather than future observations.

Uncertainty about the implementation of the chosen management alternative

There may be considerable uncertainty regarding the political and technical implementation of the assessed alternatives. This is an uncertainty that is very difficult to quantify, and it may

be advisable to at least qualitatively discuss the potential for implementation. Implementing the second best alternative is usually much better than choosing the best one if there is a considerable chance that it may finally fail in the political implementation process. Also technically, a chosen alternative may have smaller positive effects if its implementation has deficits that were not accounted for when estimating the effects. Monitoring the state of implementation should become part of the management strategy.

Making decisions based on uncertain outcomes – assessment of alternatives

Once the different sources of uncertainty have been considered to quantify the uncertainty of attributes (indicators) that describe the fulfilment of the objectives, there are several ways of coming to a decision.

The main criterion for ranking alternatives in MAVT/MAUT is maximizing the value or the expected utility among the alternatives. This requires that the trade-offs the decision-maker (society, stakeholders) is willing to make between the different objectives were quantified in the form of a multi-attribute value function. The uncertainty in attributes can then be propagated through the value function to get a probability distribution of the overall value for each alternative (under each scenario).

If the degree of uncertainty between the alternatives is large compared to the differences between alternatives, the risk attitude can be taken into account to derive a final ranking between alternatives by transforming the values into utilities. Note that – in case of risk aversion – the risk attitude will only affect the final ranking between two alternatives, if the alternative with a higher expected value has a larger uncertainty than the other (Schuwirth et al., 2012).

However, other criteria can and should be considered as well. Such criteria include:

▸ Choosing alternatives that are robust to changes under the different scenarios of socio-economic development;

▸ Choosing alternatives with high consensus potential between different stakeholders (which might disagree about the trade-offs they are willing to make between objectives) (e.g., Schuwirth et al., 2012);

▸ Applying the precautionary principle to avoid alternatives with a (quantified or unquantified) risk of unwanted outcomes;

▸ In cases where the uncertainty of absolute predictions is very high, searching for significant changes caused by the management alternatives by analysing the dependence structure of the variables contributing to overall uncertainty (Reichert and Borsuk, 2005).

Note that those criteria could be formulated as simple decision rules and applied in a qualitative way. Alternatively, they could be applied in a quantitative way by formulating them in the form of a (multi- or single attribute) value function.

In addition, the process of evaluating suggested alternatives should stimulate a creative process of generating new alternatives, e.g. through combination of promising measures from different alternatives. These additional alternatives can then be evaluated as well and may lead to a better fulfilment of the objectives or higher consensus potential. Finally, the process of generating and evaluating alternatives should continue into the future to produce an adaptive management process.

Checklist for dealing with uncertainty in AQUACROSS

The checklist is grouped around the three major sources of uncertainty identified at the beginning of this section:

1 Uncertainty about societal preferences.

   a) Were the societal preferences elicited from the relevant stakeholders or the public?

   b) Was the uncertainty of the preference quantification estimated?

2 Uncertainty about the consequences of suggested management alternatives.

   a) Were scenarios about the future socio-economic development established?

   b) Were the changes in environmental influence factors for those scenarios and their uncertainty estimated?

   c) Were the responses of ecological and economic attributes of the management alternatives estimated and their uncertainty quantified?

   d) Were these uncertainties adequately considered in the decision support process? [Consideration of risk aversion, robustness against scenarios, etc.]

3 Uncertainty about the implementation of the analysed management alternatives.

   a) Were technical or scientific risks of failure identified and were attempts made to minimise them?

   b) Was the (political) potential for implementation of the suggested management alternative estimated and considered in the recommendation?

2.6.3 Tackling multiple scales

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The meta-ecosystem approach provides a useful and powerful theoretical and conceptual tool to understand feedbacks and impacts across multiple scales and the emergent properties that arise from spatial coupling of local ecosystems, such as global source-sink constraints, biodiversity-productivity patterns, stabilisation of ecosystem processes and indirect interactions at local or regional scales. The meta-ecosystem approach thereby has the potential to integrate the perspectives of community ecology, to provide novel fundamental insights into the dynamics and functioning of ecosystems from local to global scales, and to increase our ability to predict the consequences of drivers and pressures on biodiversity and the provision of ESS to human societies.

"The problem of relating phenomena across scales is the central problem in biology and in all of science" (Levin, 1992) and problems of scale are particularly important when it comes to developing effective environmental management. There has been considerable recent attention paid to social-ecological scale mismatches and these may be observed where "human institutions do not map coherently on to the biogeophysical scale of a resource in space or time" (Cash et al., 2006). While the importance of incorporating scale considerations into environmental management has been recognised for many years (e.g., Cumming et al., 2006; Henle et al., 2010; Veldkamp et al., 2011), it remains a major practical challenge particularly when it comes to consideration of ESS (O'Higgins et al., 2010; Jordan et al., 2012). O'Higgins et al., (2014a) introduced a technique, based on the DPSWR (Driver-Pressure-State-Welfare-Response) for the identification of spatial scale mismatch. Figure 13 introduces a simple classification of scale mismatches based on the work of Cumming et al. (2006) and using the DPSWR information categories. By taking the response as the scale frame, mismatches were classified relative to it, i.e., the spatial scale of an ecological problem (comprised of pressures and states) is either larger or smaller than the fixed scale of a specific response; they classified these characteristics as grain and extent mismatches, respectively.

The AQUACROSS conceptual frame explicitly incorporates ESS within the DPSWR at the interface between State Change and Welfare. Based on attempts to map ESS values in coastal and estuarine systems (O'Higgins et al., 2010; Jordan et al., 2012), O'Higgins et al. (in review) have developed a classification of ecosystem service scale and location relationships and applied it to two estuarine CS sites. They based their classification on Fisher et al. (2009) who identified three categories of spatial relationships between ecosystem service supply (Production, P) and demand (Benefit, B); **in situ**, where P and B are co-located (e.g., a localised crab or lobster pot fishery); **omni-directional** services where P occurs in a discrete location but B is diffuse (e.g., Carbon sequestration) and **directional** where the P is in one location but B occurs in another, (e.g., the flood protection service provided by mangroves). Fisher et al. (2009) also suggested the scale qualifiers, local, regional and global for spatial characteristics and recognised the binary distinction that P and B may occur in the same place ($P_{xy} = B_{xy}$), *in*

situ, or that production and benefits may occur in different places ($P_{xy} \neq B_{xy}$): directional. They also recognised that supply might be spatially discrete but the benefits occur all around (omni–directional) implicitly recognising that P and B for some services exist on different scales.

Figure 13: DPSWR elements and scale
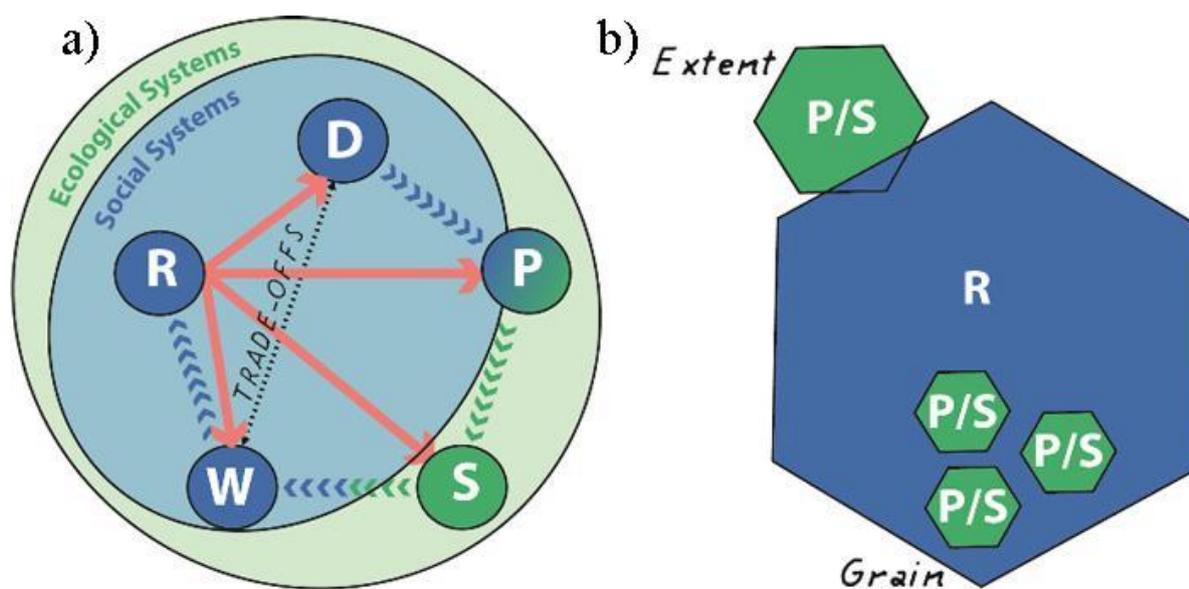
Source: O'Higgins et al., 2014a

Legend: a) Illustration of DPSWR framework showing the trade–off between the drivers of environmental state change and the changes in welfare caused by environmental change. See text for description of the DPSWR elements. b) A classification of scale mismatch. Extent mismatches occur when the pressure and state change lie partially or entirely outside the spatial domain of the response; grain mismatches occur when the spatial scale of the pressures is at too small a scale to be effectively managed by a response mechanism.

The typology below considers two distinct spatial characteristics for ecosystem service production (P) and delivery of benefits (B), those of location and scale, denoted with subscripts, XY and Z respectively. In this classification scheme spatial scale is explicitly included as a descriptor of supply and demand. In terms of spatial scale there are three possibilities, scales may be matched ($P_z=B_z$), or scales may differ ($P_z>B_z$ or $P_z<B_z$). Figure 14 summarises the six possible unique combinations of location and scale relations with a suggested names for each type of relationship.

Combining the spatial mismatch classification along with the spatial typology of ESS, there are potential design management instruments and institutions at the appropriate spatial scales for the management of ESS.

Temporal mismatches in scale can also occur, with policy objectives being set at different temporal scales than those of natural processes. Using a similar idea to those described

above, O'Higgins et al (2014b) also developed a classification of temporal scale phenomena including legacy and future effects as well as committed behaviours, which is summarised in Figure 15.

Figure 14: Typology of ecosystem services based on the location and scale

Source: O'Higgins et al., 2014a
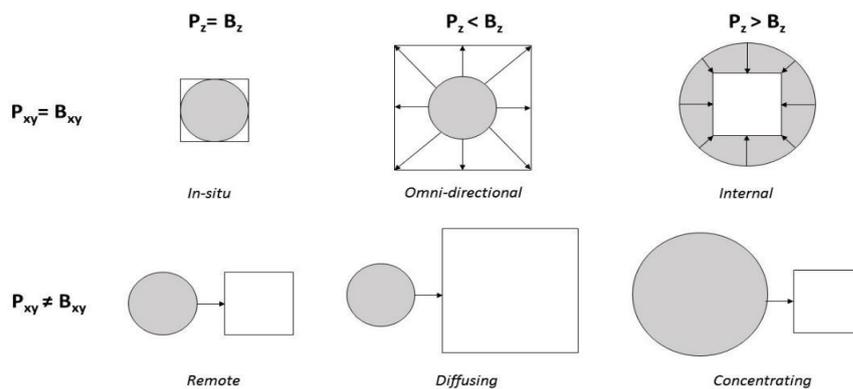
Legend: Proposed typology of ecosystem services based on the location (XY) and scale (z) of ecosystem service production (P, grey circles) and delivery of benefits (B, white squares) for each of the possible combinations of location and scale

Figure 15: Temporal scale effects
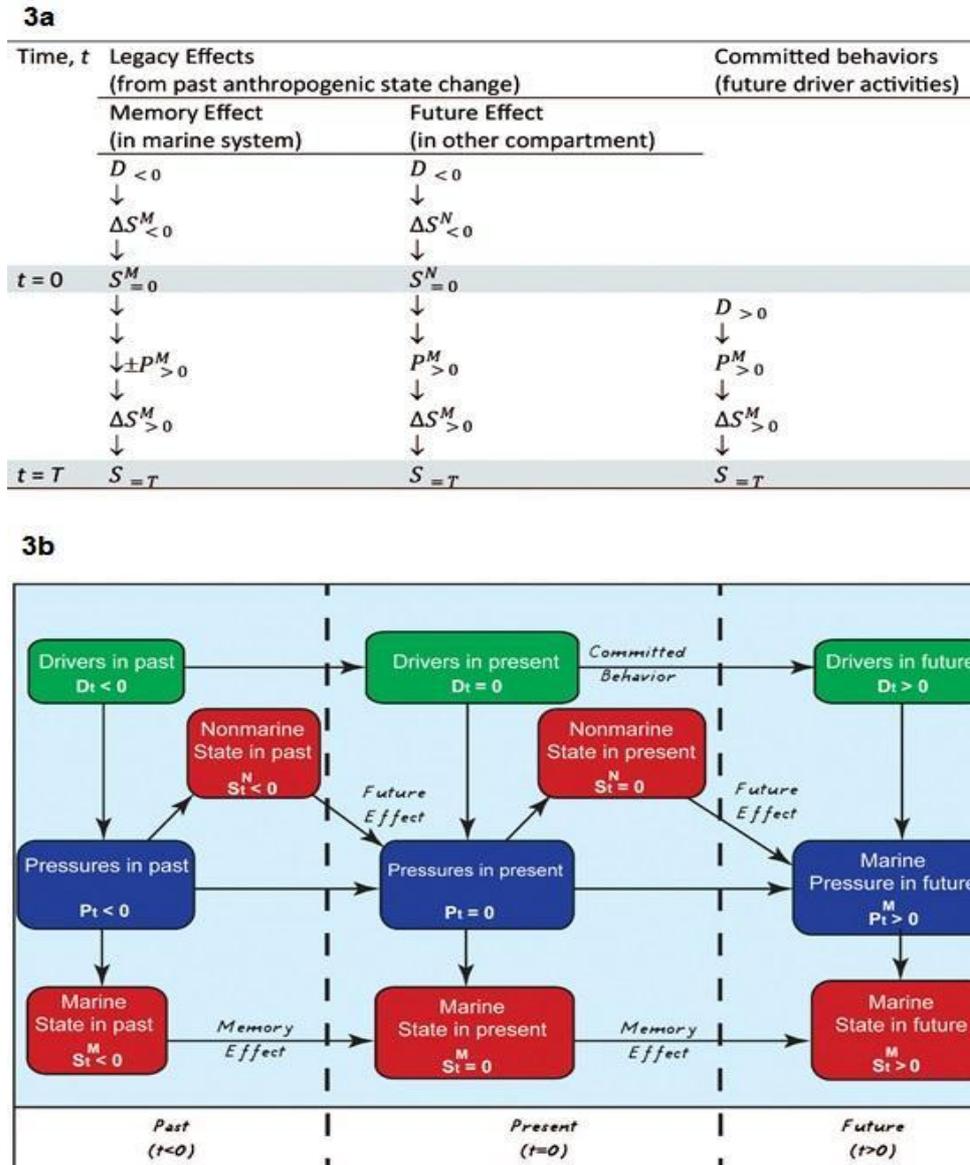
Source: O´Higgins et al (2014b)

Legend: a: t = 0 represents the time at which a decision is to be made, and t = T represents the planning horizon, so that the planning period covers the range 0 < t ≤ T. This figure adopts a similar approach to summarise the definition of the other class of endogenous constraints: Committed Behaviours showing causal relationships among a specific Driver activity (D), Pressure (P), and State (S) or State change (ΔS). Where relevant, the superscript denotes the affected ecosystem compartment, with M = marine system and N = other (non–marine) ecosystem compartments. The subscript indicates the time at which the relationship is manifested relative to the time at which a decision is made (t = 0) and to the planning horizon (t = T). b: Schematic diagram of Legacy Effects and Committed Behaviours, showing Drivers in green, Pressures in blue, and State in red.

Combining the three analytical tools summarised above and testing them in the application of case studies in the AQUACROSS project, will facilitate and enable a standardised approach to consideration of scale within the project.

Figure 16: Scale-related concepts on DPSWR cycle
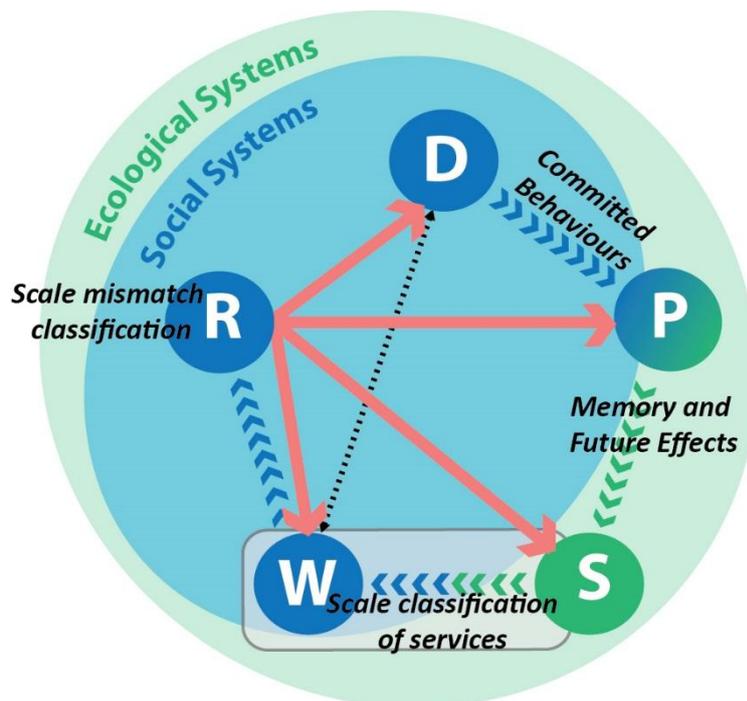
Source: O'Higgins et al., 2014a

Illustration on how the concepts relating to scale described above map on the DPSWR cycle.

By comparing Figure 16 above with Figure 12, which illustrates the location of uncertainties along the DPSWR path, it can be seen that qualitative analytical scaling tools can be used to inform assessment of uncertainties to frame various aspect of particular case studies. For example committed behaviours are defined as "collective norms and activities that are not socially or politically feasible to alter in the short to medium term" (O'Higgins et al., 2014b) and may be further categorised as (1) explicit social/political decisions that have been made prior to the planning period, effectively establishing a contract with agents such as firms whose actions have been based on these decisions, and (2) the methods for meeting demands for goods and services implicit in the operation of economic systems (O'Higgins et al., 2014b). In terms of individual AQUACROSS case studies such committed behaviours might include long standing resource management policies which are unlikely to undergo major alterations within medium term time horizons. These include the Common Fisheries and Agricultural Policies which can inform the framing of scenarios, and consideration of uncertainty around likely future socio-economic development.

Uncertainties in the future environmental forcing factors that will influence the functioning of the ecolgical system can also be reduced by explicit consideration of memory and future effects, which should be incorporated into modelled scenarios.

The effectiveness of proposed management effort of EBM strategies maybe subject to, and therefore should also account for, mismatches between scales of ecological process and scale of effective governance. While there are clear extent mismatches built into AQUACROSS case studies involving transboundary problems (for example in case studies 2 (Intercontinental Biosphere Reserve of the Mediterranean) and case study 4 (Transboundary Management of Invasive Species in Lough Erne), a systematic approach toward the analysis of scale across case studies may shed light on common challenges and solutions and the design of potential EBM response should also account for the difficulties of practical implementation of policies.

3   The Way Ahead

Unlike other projects, AQUACROSS aims to continuously review and refine this Assessment Framework towards Deliverable 3.3 (Final Assessment Framework), an updated and proof-tested output of the project, to be presented at the Final AQUACROSS Forum to be held in Berlin (Germany). For that purpose, the development of the AQUACROSS Assessment Framework you have just read through and, therefore, the investigation into the specific elements for assessment, is (i) mindful of the practical challenges to be faced in terms of applicability (e.g., linking policy and science in the three aquatic realms); (ii) makes the most out of existing knowledge to enhance current EBM practice; and (iii) ensures relevance.

More specifically, the next task under this work package will work (towards the end of the project) on the update and upgrade this Assessment Framework based on feedback from applied work in other work packages and case studies (for which ad-hoc 'cookbooks' will be developed to account for specificities and provide guidance). Project partners will thus use insights from applied work to update and revise the AF based on findings and experiences from AQUACROSS – needless to say that this process of upgrading and updating will also factor in input from stakeholders and the SPBTT.

Since the AF is a key output of AQUACROSS, special emphasis will be placed on the practical applicability of the framework in science, policy and business. In other words, significant effort will now be made to create a workable analytical framework that can be both conveyed and understood (i.e., hence using clear language and structure) as long as being flexible (incorporating varying end-user needs).

It is important to stress upon the fact that the AQUACROSS concept and AF will be applied in case studies to test and refine its applicability, thus providing the basis for its integration in the design and implementation of EBM. For this purpose, the analysis of links between drivers and pressures will feed the analysis of causalities (between biodiversity and ecological functions and services), the modelling of social-ecological dynamics, and the development of EBM.

This highlights the relevance of ulterior efforts in the project to shed further light on the link between the analysis of drivers and pressures (demand-side analysis) and the analysis of causal relationships between biodiversity, ecosystem functions, and ecosystem services (supply-side analysis, as above). This, of course, requires the tdesign, implementation, and adequate maintenance of a fully operational information system for the project, something that was considered from the onset.

Building on the overaching framework developed in this document, different modelling approaches and analytical tools will be used to evaluate the projected changes of drivers and pressures, as well as the social-ecological outcomes of those shifts, and to design new policy responses (based on an ecosystem approach) – the main value of which should neither be novelty, nor even innovation, but rather meaningfulness (i.e., actual impacts in terms of

societal challenges identified around biodiversity consevration and ESS provision).

As a result of the implementation of the AF in next stages of the project, one should be able to ascertain a number of issues:

▸ (Overall) How to move away from purely descriptive approaches towards more analytical ones, so that measuring, informing or listing, are perceived just as a what they actually are: means to a critical end (understanding, explaining, assessing to improve decision-making for enhanced levels of biodiversity and ESS delivery).

▸ How the most relevant drivers (of ecosystem change) affect aquatic ecosystems.

▸ How the demand for ecosystem services and abiotic outputs from freshwater, coastal and marine ecosystems can be met (in a sustainable way).

▸ How knowledge on biodiversity loss, drivers and indicators can be adapted, downscaled, and made useful for specific applied assessments, in the project case studies and elsewhere (once the project findings have been effectively uptaken).

▸ How the assessment of changes in the state of aquatic ecosystems can shed light on the connection between the analysis of drivers and pressures, and the ecological assessment of links between ecosystem functions, services and biodiversity.

▸ How better cases and storylines could be built and on the basis of what evidence) for biodiversity conservation and enhancement.

▸ How to underline the critical differences between causality and correlation, prediction and forecasting, statistical analysis and scientific knowledge and, in a more specific way, how to progress from predictive models towards better decision-support tools, among other things to analyse and not just measure uncertainty, a critical dimension of policy making.

▸ How to ensure that current and future models and policy-making frameworks address ecosystem-based management.

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