DESSIN

DEMONSTRATE ECOSYSTEM SERVICES ENABLING INNOVATION IN THE WATER SECTOR

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D11.1 – Internal state of the art report on ecosystem services evaluation

Water status related changes in ESS, economic valuation and sustainability assessment of ESS

Lead Author: Ecologic Institute, December 2014

The research leading to these results has received funding from the European Union Seventh Framework Programme (FP7/2007-2013) under grant agreement no. 619039

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D11.1: INTERNAL STATE OF THE ART REPORT ON ECOSYSTEM SERVICES EVALUATION	
	SUMMARY

This document provides a state-of-the-art review to support the conceptual and practical development of the DESSIN ESS Evaluation Framework. It briefly presents the state of affairs regarding the measurement of changes in ESS, including description of existing classification systems, analytical frameworks and economic valuation methodologies. The challenges associated with spatial and temporal variations of ESS are considered and an approach to define and measure sustainability is presented. Finally, the next steps in the work plan of Work Area 1 are outlined.

DELIVERABLE NUMBER	WORK PACKAGE
11.1	WP11
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ACKNOWLEDGEMENTS

The lead authors would like to thank the members of the ESE Action Group of the European Innovation Partnership on Water for their valuable input to several chapters of this document.

QUALITY ASSURANCE

Internal	First two lead authors, reviewed by all contributing authors.
PLANNED DELIVERY DATE	**ACTUAL DELIVERY DATE**
30/09/2014	01/12/2014
DISSEMINATION LEVEL	☐ CO = Confidential, only for members of the consortium

The research leading to these results has received funding from the European Union Seventh Framework Programme (FP7/2007-2013) under grant agreement no. 619039

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List of Acronyms and Abbreviations

CBA	Cost-Benefit Analysis
CICES	Common International Classification of Ecosystem Services
CIF	Common Implementation Framework
DPSIR	Drivers-Pressures-State-Impact-Responses
DSS	Decision Support System
EBM	Ecosystem-Based Management
EEA	European Environmental Agency
ESA	Ecosystem Services Approach
ESS	Ecosystem Services
EU	European Union
MA	Millennium Ecosystem Assessment
MAES	Mapping and Assessment of Ecosystem Services
MSFD	Marine Strategy Framework Directive
TEEB	The Economics of Ecosystems and Biodiversity
WA1	Work Area 1
WFD	Water Framework Directive

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Executive Summary

Global political agendas are increasingly recognising the significant contributions of freshwater ecosystems to our societies and economies. In regions like the EU, this contribution is becoming fundamental to inform management decisions concerning water resources. At present, different regions in Europe face great challenges regarding water resource quality and scarcity. While the efforts undertaken as part of the implementation process of the EU Water Framework Directive (WFD) (2000/60/EC) have brought a great deal of progress in this realm, new horizons in the region's economic and environmental policy are leading to the pursuit of common objectives through integrated action. This has put technological innovations for the water sector high on the political agenda, and has created a necessity to increase the capabilities of science to assess the full range of possible impacts of these technologies on natural systems and, subsequently, on the benefits that humans perceive from their interaction with such systems.

In this context, academic and policy communities at the European level and beyond have shown increased interest in the concept of Ecosystem Services (ESS) and the Ecosystem Services Approach (ESA). In this document, ESS are understood as the direct or indirect contributions that ecosystems make to human well-being, while the ESA is seen as the holistic perspective that includes humans, their activities and the services that ecosystems provide to humans as an integral part of the ecosystem.

As a European research project, DESSIN has the objective of operationalising the ESA to enable an extended, standardised evaluation of impacts from water-sector innovations, in particular by integrating economic, environmental and societal dimensions. Using the ESA to compare the potential of technologies may help demonstrate their additional benefits and create further incentives for market uptake. Through the development of an ESS Evaluation Framework, Work Area 1 (WA1) of the DESSIN project seeks to bridge the gap between the current evaluation capabilities and an extended assessment that describes the changes in ESS that may result from the implementation of new technologies. This document sets the foundations for the conceptual and practical development of such a framework.

The lack of agreed upon definitions and terminology revolving around ESS and their respective components hinders further research and slows progress on this topic. As a first step, this document presents a consolidated list of agreed definitions based on relevant literature. This will allow the DESSIN project to ensure coherency and a smooth understanding of ideas and concepts presented in the following work. Similarly, it is necessary to establish a conceptual approach and common classification of ESS to adopt within DESSIN's ESS Evaluation Framework. To this end, the second step provides the context of European research on ESS and the ESA and presents the CICES/MAES classification systems to be built upon within DESSIN. Furthermore, an adapted version of the well-known Driver, Pressure, State, Impact, Response (DPSIR) scheme used by the EEA, with special features to account for ESS was applied.

As there is little evidence which links changes in water status to changes in ESS, this document lays out suggested criteria to identify and select indicators/proxies to investigate the links between the two. Furthermore, the authors propose to expand the current state of knowledge on the basis of selected case studies derived from a literature review of relevant projects (e.g. FP7 DESSIN, FP7 MARS, FP7 OpenNESS). This selection must feed into a larger analytical framework which evaluates changes in ESS. Most accepted ESS assessment frameworks lack this ability to identify how changes in ecosystems will impact ESS and their provision. This document reviews a selection of alternative frameworks, concepts and methodologies which incorporate this link. Their respective advantages and disadvantages in detecting changes in ESS are discussed. Additionally, an overview of challenges associated with measuring ESS arising from spatial and temporal variations is presented.

In conjunction with the analytical framework, the ESS Evaluation Framework will also incorporate an economic valuation of ESS. This document reviews existing economic valuation methods, their strengths and weaknesses, as well as their practical application within the water sector. Ultimately, this economic valuation aims to account for values associated with societal preferences for changes in the provision of water-related ESS.

As there is an intrinsic complexity to the assessment of benefits and possible trade-offs for the implementation of new water technologies also from the perspective of sustainability, the DESSIN ESS Evaluation Framework will be supplemented with a sustainability assessment tool to identify the possible effects on different dimensions of sustainability (social, environmental and economic). This tool, based on the TRUST model, is presented to complement the ESS analytical and economic valuation components.

Finally, the next steps towards the elaboration of the DESSIN ESS Evaluation Framework are presented and briefly described. These include the identification and selection of suitable indicators/proxies to measure ecological status and their relation to ESS, the selection of an analytical framework most appropriate for the objectives of DESSIN, and the preparation of a draft evaluation methodology to be tested and refined in the near future.

Foreword

New solutions and advances in technology are needed to meet the water quality and scarcity challenges currently faced in Europe. However, such innovations are themselves typically confronted with great barriers to their implementation. One of these barriers is the difficulty of conducting comprehensive comparisons between the value of established technologies/management options and novel alternatives. Uptake is further hindered by the complexities involved in conducting assessments that consider both environmental and economic aspects when evaluating the costs and benefits of investing in and applying novel solutions. In this context, Work Area 1 (WA1) of DESSIN has the main objective of developing a tested ecosystem services (ESS) evaluation framework to estimate the potential impact of innovative technologies on freshwater ESS.

In the context of DESSIN, the Ecosystem Services Approach (ESA) may enable an extended, standardised evaluation of impacts from innovations, in particular by integrating economic, environmental and societal dimensions. Using the ESA to compare the potential of technologies may help demonstrate their additional benefits, creating further incentives for market uptake, differentiating the value propositions of the DESSIN technologies and ultimately promoting practical implementation of water-sector innovations.

The value of intact and stable ecosystems to society and their contribution to human well-being is currently a matter of critical discussion in scientific and policy circles. This has driven recent attempts in exploring and demonstrating ways to preserve and sustain natural systems. On the other hand, the current demographic and economic conditions at the global level are subjecting both public and private sectors to increased demand for goods and services in markets that are ever more competitive, which can create pressure to increase the use of natural resources. The need to balance economic growth with the protection of natural resources has traditionally been accomplished through the use of environmental legislation and regulations. These environmental policies are often viewed as a cost to economic growth with the potential to reduce economic competitiveness. However, the ESA argues that the protection of natural resources ensures the continued provision of ESS with significant benefits to society. Better understanding of these services and their accompanying benefits can help foster the research and further uptake of innovative solutions that complement and sustain existing ESS. The counterfactual is that existing decision-making frameworks that do not consider ESS are limited and obsolete (e.g. financial Cost-Benefit Analysis of alternative compliance options), as they fail to account for the full range of possible benefits.

The ESS Evaluation Framework being developed under WA1 aims to enable the assessment of changes in the provision of ESS relative to the implementation of new water technologies. The framework is being designed to operate using a common set of ecosystem typologies and ecosystem service categories (MAES/CICES). Indicators/proxies are being sought and considered that not only illustrate the linkages between the properties and functions of freshwater ecosystems and changes in ESS provision, but allow for subsequent economic valuation of these changes. Furthermore, the design of the ESS Evaluation Framework is supplemented with a sustainability assessment module.

Through its testing and application on mature case study sites within DESSIN, a beta version of the Evaluation Framework will be fine-tuned and validated. The finalised version will then be developed into a software module that will be compatible with an existing decision support system. This key outcome of DESSIN is also expected to enhance the accounting of benefits from the implementation of the WFD Programmes of Measures. Furthermore, it should help catalyse the uptake of innovative technologies in the European water sector and beyond.

The main aim of this document is to provide a state-of-the-art review to support the conceptual and practical development of the ESS Evaluation Framework in the second year of the DESSIN project. To this end, the current state of affairs regarding the tools and techniques that could be used for evaluating changes in ESS is presented. Frameworks to illustrate the interactions between society and the environment are described and the most accepted ecosystem typologies and ecosystem service categories are identified. Furthermore, an overview of the current discussions regarding the links between ecosystem biophysical functions and processes and the provision of ESS is given. Additional consideration is given to the discussion of spatial and temporal issues, both of which should be addressed within the ESS Evaluation Framework. The most common methods used for the economic valuation of ESS are explored and, finally, an approach to define and measure sustainability is presented. This initial state-of-the-art review is intended to inform the decisions of WA1 partners regarding which concepts will be integrated and developed further into the ESS Evaluation Framework in the coming months.

The structure of this document is as follows:

- **Glossary**
  - Presents a consolidated list of agreed definitions for the DESSIN project to ensure coherency and a smooth understanding of this document.
- **Chapter 1 Introduction: Developing the DESSIN ESS Evaluation Framework**
  - Provides the context of European research on ESS/ESA as well as the policy context in which this is embedded and introduces the conceptual approach of the DESSIN ESS Evaluation Framework.
- **Chapter 2 Linking changes in water status to changes in ESS**
  - Reviews the literature on ESS indication and lays out suggested criteria to identify and select indicators/proxies to investigate the links between changes in water status and changes in ESS.
- **Chapter 3 Analytical framework to evaluate changes in ESS**

- Reviews a selection of existing frameworks, concepts and methodologies for the ESS assessment and discusses their advantages and disadvantages in detecting changes in ESS.

- **Chapter 4 An approach for the spatial and temporal aspects of ESS provision**
  - Gives an overview of challenges associated with measuring ESS arising from the fact that many of these services are dispersed in space and/or time.
- **Chapter 5 Economic valuation of changes in ESS**
  - Reviews ways to account for values associated with societal preferences for changes in water ESS provision.
- **Chapter 6 How to incorporate sustainability into the DSS of DESSIN**
  - Presents a sustainability assessment tool based on the TRUST model to identify the possible effects of technology implementation on different dimensions of sustainability.
- **Chapter 7 Next steps**
  - Outlines the next steps in the work plan of Work Area 1.

Glossary

One of the first challenges encountered by WA1 of DESSIN is that, despite the emergence of recent efforts to standardise definitions and classifications, the literature on ESS still exhibits a general lack of consistency in the use of terms. In order to ensure coherency and a smooth understanding of this document, this section provides a list of agreed definitions for the DESSIN project. This effort to overcome the ambiguity of the terminology found in the literature will also help to inform the development of the practical ESS Evaluation Framework during the second year of the DESSIN project.

Term	Description
DPSIR	Drivers-Pressures-State-Impact-Responses The causal framework for describing the interactions between society and the environment adopted by the European Environment Agency: driving forces, pressures, states, impacts, responses (Gabrielsen and Bosch, 2003). • Driver = An anthropogenic activity that may have an environmental effect (e.g. agriculture, industry) (MARS Project Terminology, 2014) • Pressure = The direct environmental effect of the driver (e.g. an effect that causes a change in water flow or a change in the water chemistry) (MARS Project Terminology, 2014) • State = The condition of the ecosystem under study (e.g. water body) resulting from both natural and anthropogenic factors (i.e. physical, chemical and biological characteristics) • Impact = Effects on ecosystem services (Impact I) and their subsequent effects on human wellbeing (Impact II) resulting from changes in ecosystem state, triggering social *Response.* See *Impact I* and *Impact II* below (Müller and Burkhard, 2012) • Response = The measures taken to address drivers, reduce pressures and improve the state of the ecosystem under study (e.g. restricting abstraction, limiting point source discharges, developing best practice guidance for agriculture) (MARS Project Terminology, 2014) (The MARS Project Terminology, 2014 is based on IMPRESS, 2002)
Impact I	The changes in ecosystem services induced by modifications in the state of an ecosystem (based on Müller and Burkhard, 2012).
Impact II	The effects that changes in ecosystem services have on human-well-

	being and on the value of the benefits perceived from ecosystem service use (based on Müller and Burkhard, 2012).
Ecosystem	The environmental system of interest within the DESSIN project (e.g. a surface or ground water body, sub-catchment or catchment).
Ecosystem capacity	Refers to the capability of a particular area to provide a specific bundle of ecosystem goods and services within a given time period (Burkhard et al., 2012).
Ecosystem functions	Subset of the interactions between biophysical structures, biodiversity and ecosystem processes that underpin the capacity of an ecosystem to provide ecosystem services (TEEB, 2010 as in MAES Glossary of terms, 2013).
Ecosystem processes	All interactions between elements of the ecosystem. Note: Ecosystem functions represent a specific subset of the ecosystem processes (*see Ecosystem functions above*).
Ecosystem properties	Elements, structure and processes of the ecosystem (see *Ecosystem structure* and *Ecosystem processes*).
Ecosystem state	See *DPSIR – State.*
Ecosystem status	'Water status' according to the WFD. This is, the general expression of the status of a body of water as determined by the poorer of its ecological status and its chemical status (in the case of surface water) or the poorer of its quantitative status and its chemical status (in the case of ground water). Ecosystem status is a subset of the *DPSIR - State* category.
Ecosystem structure	The biophysical architecture of ecosystems (TEEB, 2010).
Ecosystem services (ESS)	The contributions that ecosystems make to human well-being. They are seen as arising from the interaction of biotic and abiotic processes, and refer specifically to the 'final' outputs or products from ecological systems (Haines-Young and Potschin, 2011).
Provisioning services	All nutritional, material and energetic outputs from living systems. In the CICES structure a distinction is made between provisioning and material outputs arising from biological or organic materials (biomass) and water (Haines-Young and Potschin, 2011).
Regulating and maintenance services	All the ways in which living organisms can mediate or moderate the ambient environment that affects human performance. It therefore covers the degradation of wastes and toxic substances by exploiting living processes. Regulation and maintenance also covers the mediation

	of flows in solids, liquids and gases that affect people's performance, as well as the ways living organisms can regulate the physico-chemical and biological environment of people (Haines-Young and Potschin, 2011).
Cultural services	All the non-material, and normally non-consumptive, outputs of ecosystems that affect physical and mental states of people. Cultural services are primarily regarded as the physical settings, locations or situations that give rise to changes in the physical or mental states of people, and whose character are fundamentally dependent on living processes; they can involve individual species, habitats and whole ecosystems. The settings can be semi-natural as well as natural settings (i.e. can include cultural landscapes) providing they are dependent on in situ living processes (Haines-Young and Potschin, 2011). • Settings for physical activities • Settings for intellectual or mental interactions • Settings for religious or spiritual activities obvious
Ecosystem Services Approach (ESA)	A holistic perspective that includes humans, their activities and the services that ecosystems provide to humans as an integral part of the ecosystem.
Ecosystem Service Profile (ESP)	The match between the societal use of ecosystem services and the provision of those services. For each individual ecosystem service, its status is defined by the provision-to-use ratio (R) (Paetzold et al., 2010). The ESP approach is a multi-criterion, context-specific assessment.
ESS provision	The actual provision of ecosystem services.
ESS use	The actual utilisation of ecosystem services by people.
Ecosystem goods and services	See *Ecosystem services (ESS).*
Ecosystem benefits	The societal gain/s related to the actual utilisation of an ecosystem service.
Human Well-being	"Human well-being is that which arises from adequate access to the basic materials for a good life needed to sustain freedom of choice and action, health, good social relations and security." The state of well-being is, among other issues, dependent on the aggregated output of ecosystem services and thus changes in the latter can affect the former (Haines-Young and Potschin, 2011).
Sustainability of ecosystem service use	Sustainability of the ecosystem service use is met when the actual use of an ecosystem service is not exceeding its capacity (Paetzold et al., 2010).

1. Introduction: Developing the DESSIN ESS Evaluation Framework

Freshwater ecosystems provide a number of valuable goods and services to society. These goods and services are underpinned by the interactions between biophysical structures, biodiversity and ecosystem processes (TEEB, 2010 as in MAES Glossary of terms, 2013). As proposed by TEEB (2010) and Haines-Young and Potschin (2013), ESS are defined as the direct and indirect contributions that ecosystems make to the well-being of society as a whole.

However, the ability of these ecosystems to provide services is not unlimited but is dependent on the type of ecosystem and its full functioning, both of which can be easily hampered by human-induced pressures. These pressures may affect ecosystem structures, habitats and/or biodiversity, thus modifying the conditions under which ecosystems operate, their capacity to provide services and the value associated with the latter.

In this context, it may be useful to consider an approach that combines understanding of ecosystem functioning with an understanding of how humans interact with and benefit from ecosystems. Ecosystem-Based Management (EBM)[1] and the ESA account for all the complexities of the system, moving away from a reductionist approach which focuses on individual ecological components, pressures or sectors to a more holistic view that includes humans, their activities and the services that ecosystems provide to humans as an integral part of the ecosystem.

Justification for using the ESA to inform decisions about the management of water resources

With the emergence of modern environmentalism in the second half of the 20th century, specialised economic disciplines, like environmental and resource economics, started to address shortcomings in standard economic science to analyse environmental problems (Røpke, 2004). From the 1970s on, a series of contributions started referring to the way particular functions of nature serve human societies and a growing number of authors started to frame ecological concerns in economic terms in order to stress societal dependence on natural ecosystems, representing the origins of the modern ESA (Gómez-Baggethun et al., 2010). The recognition of the role of nature in supporting the economy and human well-being also motivated incorporation of the ESA into existing decision-making frameworks, such as cost-benefit analysis. As early as 1977, Westman (as cited in Fisher et al. (2009)) suggested that the social value of the benefits ecosystems provide could potentially be enumerated so that society could make more informed policy and management decisions. Over time, the need for an ESA to natural resources management has been recognised in policy, from an international level to regional and national levels worldwide.

[1] „Ecosystem-Based Management is an adaptive, learning-based process that applies the principles of the scientific method to the process of management." EBM is similar to the ESA in that at its core is the recognition of humans and their activities as integral parts of the ecosystem. It is also concerned with the processes underlying living systems and with the continuity of ecosystem goods and services provision (UNEP/GPA, 2006).

According to Fisher et al. (2008) some indications of the potential of integrating the ESA into natural resources management schemes are:

- Managing for well-functioning ecosystems provides services more cheaply and more reliably than typical built–capital responses (Van Wilgen et al., 2004; Dietz et al., 2003)
- Economic values of ESS gathered through research can support the argument for the inclusion of the ESS notion in regional land and water use decision-making
- Ecosystem service research can be designed to have strong policy foresight, stimulate cooperation between a broad range of policy agents and scientists, and induce significant effects
- Raising public awareness of the importance of well-functioning ecosystems can be facilitated by the expression of this importance in economic terms

In addition to the outlined academic exercises investigating the relationship between biodiversity, ESS and the ESA, numerous ongoing research projects are being conducted on these topics. In order to position our work within this framework, Table 1 below highlights the most relevant European research projects focusing on ESS and the ESA.

Table 1. The context of EU research on ESS/ESA

Project or Initiative Title	Brief Description	Time Frame
Biodiversity and Ecosystem Service Sustainability **(BESS)**[2]	Research program designed to reduce uncertainty about the functional role of biodiversity in key ecosystem processes and the delivery of ecosystem processes at the landscape scale, as well as how these are likely to change in an uncertain future.	2011-2017
Urban Biodiversity and Ecosystem Services (URBES)[3]	Addresses significant scientific knowledge gaps on the role of urban biodiversity and ecosystem services for human well-being.	2011-2014
CONNECT[4] - Linking biodiversity conservation and ecosystem services	Linking biodiversity conservation and ecosystem services: advancing insights in trade-offs and synergies between biodiversity, ecosystem functioning and ecosystem service values for improved integrated biodiversity strategy.	2012-2014
RUBICODE - Rationalising Biodiversity Conservation in Dynamic Ecosystems[5]	Aiming to develop and apply concepts of dynamic ecosystems and the services they provide; explore relationships between service-providing populations, ecosystem resilience, function and health; and socio-economic and environmental drivers of biodiversity change.	2006-2009
MARS (Managing Aquatic ecosystems and	Supports European policies by helping water managers and policy-makers to understand the effects of multiple stressors on	2014-2018

[2] http://www.nerc-bess.net/
[3] http://urbesproject.org/
[4] http://www.connect-biodiversa.eu/
[5] http://www.rubicode.net/rubicode/index.html

water Resources under multiple Stress)[6]	surface waters and ground water, their biota and the services they provide to humans; to understand how ecological status and ecosystem services are related – if at all; to advise river basin managers how to restore multi-stressed rivers and lakes; to advise the revision of the Water Framework Directive on new indicators for ecological status and ecosystem services; and to develop methods and software for the Programmes of Measures.	
OPENNESS (Ecosystem services – from concepts to real-world applications)[7]	Aims to translate the concepts of Natural Capital and Ecosystem Services into operational frameworks that provide tested, practical and tailored solutions for integrating ESS into land, water and urban management and decision-making processes.	2012-2017
POLICYMIX[8]	Assesses the role of economic instruments in policy mixes for biodiversity conservation and ecosystem services provision.	2010-2014
OPERAs[9] - Ecosystem Science for Policy and Practice	An initiative to define whether, how and under what conditions the concepts of Ecosystem Services and Natural Capital can be transferred and applied to the actual management of a variety of ecosystems in order to enhance human well-being.	2012-2016
BESAFE (Biodiversity and ES, arguments for our future environment)[10]	Aims to improve our understanding of the alternative ways in which concepts for the 'value of biodiversity' can be used to improve biodiversity policy-making and governance at local, national and European to global scales.	2011-2015
BIOMOT (MOTivational strength of ecosystem services and alternative ways to express the value of BIOdiversity)[11]	An initiative to address the problem of building and sustaining motivation to act for biodiversity by means of a comprehensive rethinking of what value and motivation actually are for people.	2011-2015
BIOMES (Biodiversity, water and Ecosystem Services)[12]	An action to provide the scientific basis for the integration of policies, with a focus on mapping ESS, developing robust modelling approaches that evaluate ecosystem dynamics and simulations of future scenarios and providing scientific support to EU nature and water policies which have ecosystem-based management targets.	2012-2013
VOLANTE (Visions of land use transactions in Europe).[13]	An initiative to provide European policy and land management with clear visions on how to reduce the large variation in possible land use scenarios for the future to a manageable set.	2010-2015

[6] http://www.mars-project.eu/
[7] http://www.openness-project.eu/
[8] http://policymix.nina.no/
[9] http://operas-project.eu/operas/index.html
[10] http://www.besafe-project.net/
[11] http://www.biomotivation.eu/
[12] http://ies.jrc.ec.europa.eu/
[13] http://www.volante-project.eu/

	This vision development is enabled through the use of expertise on land use change at various spatial and temporal scales.	
EU BON (Building the European Biodiversity Observation Network)[14]	Aims to build a substantial part of the Group on Earth Observation's Biodiversity Observation Network (GEO BON). In light of the new Intergovernmental science-policy Platform on Biodiversity and Ecosystem Services (IPBES), such a network and approach are imperative for attaining efficient processes of data collation, analysis and provisioning to stakeholders.	2012-2017
ESAWADI (Utilising the Ecosystem Services Approach for Water Framework Directive Implementation)[15]	Aims to analyse and provide advice on the potential usefulness of the ESA in view to support the implementation of the EU WFD. The project partners developed a common framework of analysis, implemented case studies in France, Germany and Portugal and analysed results on the added-value of the ESA.	2010-2013

Policy context

As mentioned above, the significant contribution of the services provided by freshwater ecosystems to our societies and economies is becoming increasingly recognised in the global political agenda. In regions like the EU, this contribution is also becoming fundamental to inform future management decisions concerning water resources. Several policy initiatives now translate this challenge into actual implementation (e.g. the EU Biodiversity Strategy to 2020 (Target 2, Action 5) and the Marine Strategy Framework Directive (MSFD) (2008/56/EC)). However, freshwater policy in the EU directs the focus of management to achieving a certain ecosystem state (e.g. Good Ecological Status under the WFD (2000/60/EC)); and, thus, has no link with an ESS-oriented management perspective, as the objective of policy is not to maximise the delivery of ESS. However, the EU DG Environment has recently issued guidelines on the use of ecosystem service assessments in the implementation of the WFD (2000/60/EC) (and also the EU Floods Directive (2007/60/EC)) (Sørensen et al., 2014). One of the objectives of DESSIN, and specifically of WA1, is to explore the connection between Good Ecological Status (as defined by the WFD (2000/60/EC)) and the provision of several societal benefits.

On the front of innovation and economic growth, the European Innovation Partnership on Water (EIP Water)[16] is a recent initiative within the EU 2020 Innovation Union[17] that aims to address the major challenges faced by the European water sector. EIP Water is integrated by a number of Action Groups that serve as platforms for collaboration, pooling of expertise and the creation of synergies between public and private actors. This is expected to speed up innovations and create market opportunities for them. Within this initiative, the discussion on ESS is taken on by the Ecosystem Services for Europe Action Group (ESE AG)[18]. The main aim of the ESE AG is to develop a

[14] http://www.eubon.eu/
[15] http://www.esawadi.eu/
[16] For more information visit: http://www.eip-water.eu/about
[17] For more information visit: http://ec.europa.eu/research/innovation-union
[18] For more information visit: http://www.eip-water.eu/working-groups/ese-ecosystem-services-europe-ag052

methodology to assess ecosystem benefits. Given the common interests and synergies between the ESE AG and DESSIN, a network for exchange and collaboration between the two was established.

The need for a consistent typology of ESS

If ESS are to provide an effective framework for decision-making, they must be classified in a way that allows for comparison of trade-offs among potential benefits. This means that the full range of benefits reflecting human well-being from ecosystems must be represented in an effective typology of ESS (Wallace, 2007). The three most recognised international ESS classification systems currently available are the "Millennium Ecosystem Assessment" (MA) (2003), "The Economics of Ecosystems and Biodiversity" (TEEB) and the "Common International Classification of Ecosystem Services" (CICES) (Haines-Young and Potschin, 2011). In essence, they resemble each other to a large extent (see Table 2 below) due to the fact that these approaches build on one another.

Millennium Ecosystem Assessment (MA)[19]

The MA was the first large-scale ecosystem assessment and contributed significantly to putting ESS firmly on the policy agenda (Fisher et al., 2009). It was initiated by the United Nations and launched in 2001.

The MA was established to provide an integrated assessment of the consequences of ecosystem change for human well-being and to analyse options available to enhance the conservation of ecosystems and their contributions to meeting human needs (Millennium Ecosystem Assessment, 2003):

> *"Sound policy and management interventions can often reverse ecosystem degradation and enhance the contributions of ecosystems to human well-being, but knowing when and how to intervene requires substantial understanding of both the ecological and the social systems involved. Better information cannot guarantee improved decisions, but it is a prerequisite for sound decision-making. The Millennium Ecosystem Assessment was established to help provide the knowledge base for improved decisions and to build capacity for analyzing and supplying this information."*

The MA provides a framework that has been adopted and further refined by TEEB and CICES. This classification is globally recognised and used in sub-global assessments (Maes et al., 2013). The MA organises ESS into four groups: *provisioning services, regulating services, supporting services and cultural services*.

The Economics of Ecosystems and Biodiversity (TEEB)[20]

TEEB is a global initiative focused on drawing attention to the economic benefits of biodiversity, including the growing cost of biodiversity loss and ecosystem degradation, and was launched in response to a proposal by the G8+5 Environment Ministers (Potsdam, Germany 2007) to develop a global study on the economics of biodiversity loss.

[19] For more information visit: http://www.millenniumassessment.org/
[20] For more information visit: http://www.teebweb.org/

TEEB presents an approach that can help decision-makers recognise, demonstrate and capture the values of ESS and biodiversity. The initiative uses a typology which mainly follows the MA classification: *provisioning services, regulating services, habitat services and cultural and amenity services*.

Common International Classification Ecosystem Services (CICES)[21]

CICES is a standardised classification system developed by the European Union. The first draft of CICES was tabled for discussion in December 2009 by the European Environment Agency (EEA), and updated versions have followed since as a result of consultations with members of the different user communities (the latest version, V4.3, was published in 2013) (Haines-Young and Potschin, 2011). CICES is the core of EU efforts to develop a consistent classification of ESS for ecosystem mapping.

CICES provides a hierarchical system that builds on the MA and TEEB classifications and differentiates between *provisioning services, regulating and maintenance services and cultural services*.

Table 2. Ecosystem services categories in MA, TEEB and CICES (Maes et al., 2013)

	MA	TEEB	CICES V4.3
Provisioning services	Food (fodder)	Food	Biomass [Nutrition]
			Biomass (Materials from plants, algae and animals for agricultural use)
	Fresh water	Water	Water (for drinking purposes) [Nutrition]
			Water (for non-drinking purposes) [Materials]
	Fibre, timber	Raw Materials	Biomass (fibres and other materials from plants, algae and animals for direct use and processing)
	Genetic resources	Genetic resources	Biomass (genetic materials from all biota)
	Biochemicals	Medicinal resources	Biomass (fibres and other materials from plants, algae and animals for direct use and processing)
	Ornamental resources	Ornamental resources	Biomass (fibres and other materials from plants, algae and animals for direct use and processing)
			Biomass based energy sources

[21] For more information visit: http://cices.eu/

			Mechanical energy (animal based)
Regulating services (TEEB) Regulating and supporting services (MA) Regulating and maintenance services (CICES)	Air quality regulation	Air quality regulation	[Mediation of] gaseous/air flows
	Water purification and water treatment	Waste treatment (water purification)	Mediation [of waste, toxics and other nuisances] by biota
			Mediation [of waste, toxics and other nuisances] by ecosystems
	Water regulation	Regulation of water flows	[Mediation of] liquid flows
		Moderation of extreme events	
	Erosion regulation	Erosion prevention	[Mediation of] mass flows
	Climate regulation	Climate regulation	Atmospheric composition and climate regulation
	Soil formation (supporting service)	Maintenance of soil fertility	Soil formation and composition
	Pollination	Pollination	Lifecycle maintenance, habitat and gene pool protection
	Pest regulation	Biological control	Pest and disease control
	Disease regulation		
	Primary production Nutrient cycling (supporting services)	Maintenance of life cycles of migratory species (incl. nursery service)	Lifecycle maintenance, habitat and gene pool protection
			Soil formation and composition
			[Maintenance of] water conditions
		Maintenance of genetic diversity (especially in gene pool protection)	Lifecycle maintenance, habitat and gene pool protection
Cultural services	Spiritual and religious values	Spiritual experience	Spiritual and/or emblematic
	Aesthetic values	Aesthetic information	Intellectual and representational interactions

	Cultural diversity	Inspiration for culture, art and design	Intellectual and representational interactions
			Spiritual and/or emblematic
	Recreation and ecotourism	Recreation and tourism	Physical and experiential interactions
	Knowledge systems and educational values	Information for cognitive development	Intellectual and representational interactions
			Other cultural outputs (existence, bequest)

Policy-led initiatives: Working Group MAES[22]

The Working Group on Mapping and Assessment on Ecosystems and their Services (MAES) was established under the Common Implementation Framework (CIF) to support the implementation of Action 5 of the EU 2020 Biodiversity Strategy (referred to above). The group is composed of Member State representatives, scientific experts, the EEA and EU staff members. The WG MAES works on a conceptual framework for mapping and assessing the link between human well-being and biodiversity. It has also put forward proposals for a typology of ecosystems and ESS based on CICES. The MAES framework, because of its numerous features and possible applications, is slowly becoming the reference point in policy evaluations about ecosystem service assessments. Its main features are:

- Identifies human benefits from ecosystem goods and services and separates benefits from economic values
- Identifies the services from ecosystems and biodiversity that are actually used by humans and which enhance human well-being
- Highlights that ESS stem from the ecological structure and processes and their functions in ecosystems
- Illustrates that there is a feedback via pressures (drivers of change), including pressure-mitigating policies (responses)
- Establishes feedbacks via institutions, judgments, management and restoration, which connects the social sciences angle with the natural sciences angle to ESS

The conceptual approach for an assessment framework to evaluate changes in ecosystems services relevant to the water sector

Given the fact that DESSIN is embedded within the context of the EU, aligning our efforts to those of WG MAES and building upon their progress is seen as a natural step by the partners in WA1. For this reason, it has been decided to adopt the MAES/CICES ecosystem typologies and ecosystem service categories for the development of the DESSIN ESS Evaluation Framework.

[22] Mapping and Assessment of Ecosystem Services, for more information visit http://biodiversity.europa.eu/maes

Furthermore, the Framework is being designed on the basis of the well known *Driver, Pressure, State, Impact, Response (DPSIR)* scheme used by the EEA, with special features to account for ESS. Specifically, the proposed concept comprises an adaptation of the approaches by Müller and Burkhard (2012) and van Oudenhoven et al. (2012), which in turn follow the "ecosystem service cascade" of Haines-Young and Potschin (2010; 2013) also used in Maes et al. (2013). Besides making reference to the DPSIR elements, this approach shows the linkages between environmental state descriptions (ecosystems and biodiversity) and human systems (human well-being) as a part of the adaptive management cycle. This approach is considered by WA1 partners to be the one that can best be adapted to the needs of DESSIN.

Figure 1 outlines the DPSIR scheme as applied in DESSIN. In it, the innovative technologies to be trialled within the project are considered *Responses* that may have impacts on *Drivers* (anthropogenic activities with environmental impacts), *Pressures* (the direct effects of such activities) and *States* (the conditions of the ecosystems under study). From the resulting changes in ecosystem state, the changes in ESS (Impact I) will be estimated. An economic assessment of the subsequent changes in the benefits perceived by society and in the value of the goods and services derived from ecosystems (Impact II) will follow. Finally, this estimated change in the level of human well-being will provide insights for the conduction of a sustainability assessment to inform policy and decision-making (further responses).

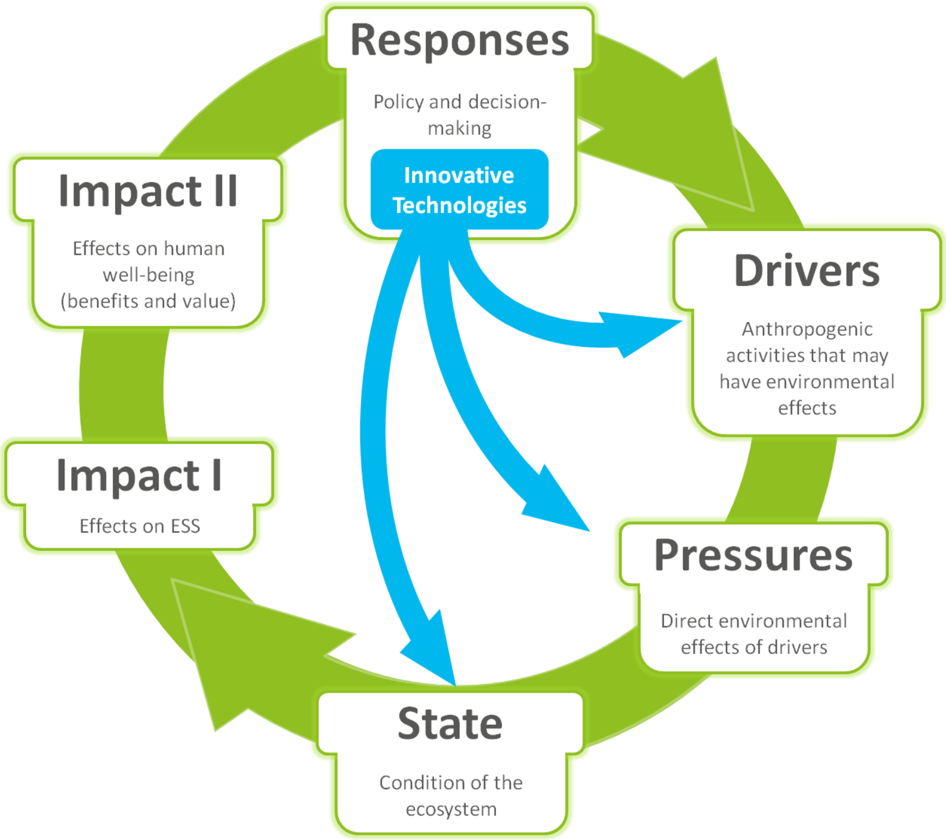

Figure 1. Conceptual approach of the DESSIN ESS Evaluation Framework (based on Müller and Burkhard, 2012, Van Oudenhoven et al., 2012 and Haines-Young and Potschin, 2010; 2013)

2. Linking changes in water status to changes in ESS

Faced against the substantial scientific and administrative challenges posed by the requirements to implement the EU WFD, an ecosystem service-based approach (i.e. ESA) has been proposed to support the implementation process (Vlachopoulou et al., 2014; Sørensen et al., 2014). The ESA is expected to contribute to a more transparent and participatory understanding of the WFD's objectives, and to introduce a more holistic perspective including the consideration of wider benefits beyond the immediate water environment (Blackstock et al., in press). Operationalising this theory, however, requires careful evaluation of the synergies and potential conflicts of the aims and objectives of the WFD (2000/60/EC) and the ESA.

Here, we discuss the relationship between the WFD and the ESA and compare their objectives with the aims in DESSIN, i.e., to demonstrate a methodology for the valuation of ESS promoting innovation in water management. The main objective of the WFD is to reach good water status, comprising specific quantitative and qualitative criteria such as ecological status. By pursuing an indicator-oriented, biophysical perspective we first compare the basic features of both the WFD (2000/60/EC) and the ESA and review existing knowledge on linking ecological status and services. Referring to the adaptive DPSIR management cycle (EEA, 1999) allows us to refine the original research question on linking changes in water status to changes in ESS, asking more specifically about the relationship between the indicators used to measure ecological status and ESS. Our work reveals certain flaws in the in the current indication of ESS: besides a common conceptual fuzziness with putative service indicators in fact addressing driving factors, pressures or states, empirical linkage between ecosystem properties, functions and ESS is often lacking. The distinction between ecosystem structure-related and ecosystem process-related services, however, allows for a first orientation regarding suitable indicator selection.

We understand 'ecosystem properties' here (following van Oudenhoven et al., 2012) as comprising ecosystem components (or elements, to follow the terminology of the WFD (2000/60/EC)), structures and processes. ESS are here understood as 'the contributions that ecosystems (whether natural or semi-natural) make to human well-being; their fundamental characteristic is that they retain the link to underlying ecosystem functions, processes and structures.' These services (often together with human labour) then feed into benefits to humans. A fish that can be harvested would thus be the ecosystem service while food made from this would be the benefit. Water purification would be the service, while clean drinking or bathing water would be the benefit. It is also important to distinguish the actual flow or delivery of ESS from the potential of the ecosystem to deliver them. What we here call ecosystem service potential is often also labelled "function". According to de Groot et al. (2002) these functions encompass (i) regulation of essential ecological processes and life support systems through bio-geochemical cycles and other biospheric processes and production processes, (ii) habitat providing refuge and reproduction spaces to living organisms, (iii) production (e.g. photosynthesis) and (iv) information providing opportunities for reflection, spiritual enrichment, cognitive development, recreation and aesthetic experience.

According to Bauer and Johnston (2013), the "current enthusiasm for the ecosystem service concept has led to numerous empirical applications that sacrifice scientific rigor in ways that provide imprecise or misleading information." These authors notice a trade-off of purism versus practicality that challenges a plausible and reliable application of the concept. Our contribution is meant to strengthen the scientific rigor of applying the Ecosystem Approach, focussing on its biophysical foundations. In particular, it guides the selection of service indicators, especially relevant for the meaningful quantification of ESS.

Comparing WFD and ESA

Table 3 summarises the basic features of the two frameworks regarding their perspectives, underlying values, management objectives and spatial scales. While the WFD (2000/60/EC) represents a statutory framework of formalised design, the ESA is more flexible, with the actual setup depending on the particular application purpose in focus and the services relevant to it. Both concepts are supposed to be implicitly connected, but their underlying values are principally different: non-human nature is valued by the WFD, with the natural ecosystem state undisturbed by man being the reference condition. In contrast, the ESA is oriented towards human benefits and well-being that depend on ESS. Their sustainable provision represents the central objective for managing the ecosystem. The WFD (2000/60/EC) defines the concrete objective of achieving good ecological and chemical status of the water body. Thus, the spatial scale addressed by the WFD is more definite and constrained by the natural boundaries of the river basin. The target of the ESA is more open and strongly subject to specific societal contexts. The relevant spatial scales within the ESA can vary from local (e.g. provision of fish) to global (e.g. carbon sequestration). Further distinction between the locations of service generation and service use further complicate distinct spatial allocation (Hein et al., 2006).

In summary, both concepts hold complementary potential but show many distinct features. This poses challenges in operationalising the ESA in the WFD (2000/60/EC) context. Linking changes in water status to changes in ESS allows for investigating into the practical consequences of merging both concepts.

Table 3. Comparison between the basic features of the Water Framework Directive (WFD) and the ecosystem service-based approach (ESA)

	WFD	ESA
Perspective	Specific ecosystem properties	Selected ecosystem services*
Value	Orientation by "natural" state	Human welfare and benefit
Management Objective	Good water body (i.e. chemical and ecological) status	(Sustainable) service provision
Spatial Scale	Water body (within catchment)	Variable (local to global)

* Choice depends on contextual values

Existing knowledge on linking ecological status and services

While we are focussing here on specific empirical relations between ecological status and ecosystem service provision, at the general level a potential mismatch is already clear between the targets followed by both approaches. Ecological status is by definition best without human uses, but the ESA by definition focuses on the use of ecosystems by humans. Although the potential of ecosystems to deliver services (e.g. for water purification) may be best in near to natural ecosystems, the actual use of ESS flows (and thus the tapping of this potential by e.g. input of sewage) may compromise this very potential and thus good ecological status. An important question is thus, not just to ask for a maximum of ESS provision but for how to arrive at a sustainable delivery of these services that can keep up the potential in the long run. It has to be scrutinised if and under what circumstance (e.g. for which services) this is possible while at the same time maintaining good ecological status.

In the literature, it is often assumed that a positive linear relationship exists between ecological status and ESS (e.g. Keeler et al., 2012). In theory, the link between these two concepts can either be existent or non-existent. If linkage exists, it can either be linear with positive or negative sign, or non-linear (unimodal, multimodal, threshold function, etc.).

Prime examples of empirical relationships are given for services relating to fish provision. Tolonen et al. (2014) recently evaluated the relevance of ecological status to ESS in a large boreal lake. They correlated the ecological assessment score of different sub-basins of the Finnish lake Päijänne with fish catches and reproductive potentials of whitefish, demonstrating positive relationships. Similar findings are reported by Ludsin et al. (2001) that observed a recovery of intolerant species desired in sport or commercial fisheries following phosphorus abatement programmes at Lake Erie. Hartmann and Quoss (1993) and Stockner et al. (2000) described markedly opposite relationships, with lake oligotrophication linked to decreasing productivity of commercially relevant fish. These examples demonstrate existing relationships between ecological status and selected ESS, but also highlight the ambivalent nature of these relationships, mainly depending on case-specific value preferences (e.g. ecological characteristics of the fish species demanded by the consumer).

The conceptual studies of Braat and de Groot (2012) or Kandziora et al. (2013) propose mainly different relationships of provisioning, regulating and cultural services to ecosystem state (Figure 2). While the provisioning services generally show positive correlation with increasing ecosystem degradation, regulating services are negatively related. Cultural services are assumed to feature a unimodal distribution along the gradient of ecosystem state, peaking at conditions of slight ecosystem impairment. These schemes offer a conceptual framework linking ecological status to different kinds of ecosystem services. However, as demonstrated in the examples given above, actual linkages seem highly specific and require empirical investigation on the basis of individual case studies.

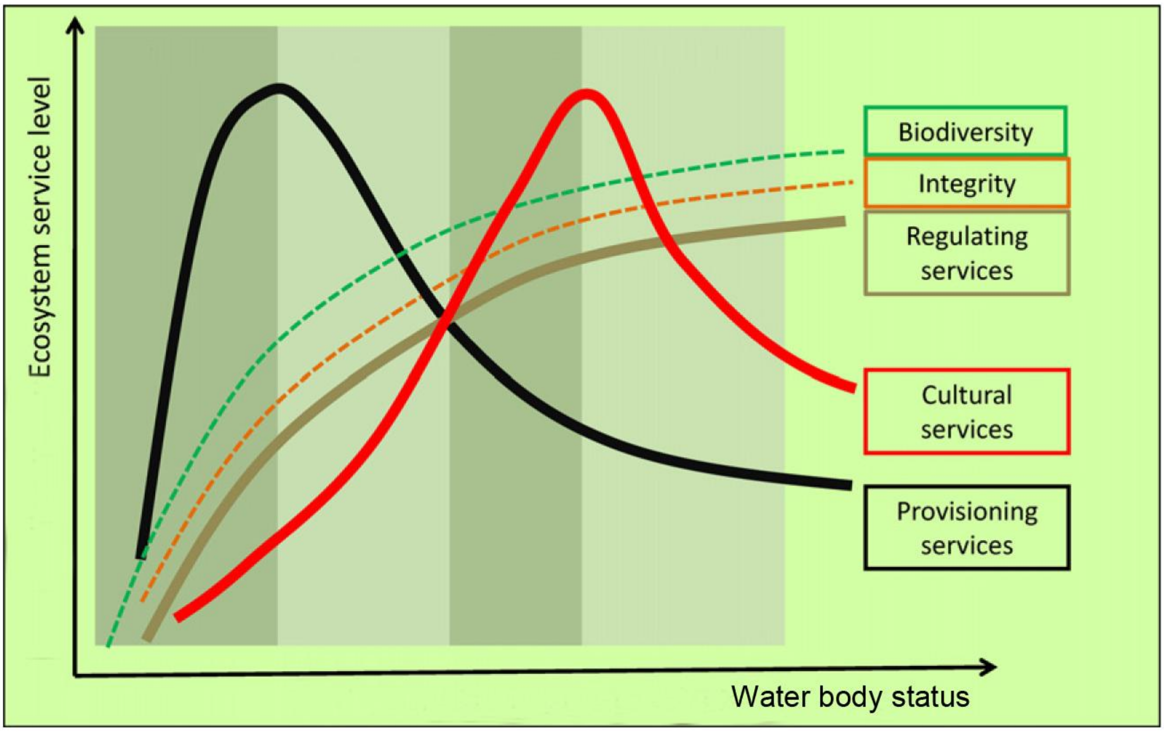

Figure 2. Hypothetical relationship between water body status and service provision (Kandziora et al., 2013; modified)

How do state and performance indicators relate?

The DPSIR model allows to position the two aspects of ecosystem state and services in an established conceptual framework (Figure 1). Any empirical investigation of the relationship can start with the analysis of indicators quantifying the ecological status ('state indicators') and ESS ('performance indicators'; van Oudenhoven et al., 2012).

Indicators used to measure ecological status

The ecological status of Europe's surface waters is measured on the basis of selected ecosystem properties, specifically ecosystem elements. EU Member States developed about 300 different assessment methods addressing the composition and abundance of various biological elements such as aquatic flora, invertebrates and fishes (Birk et al., 2012). These national methods were demonstrated to yield comparable results via an extensive intercalibration exercise (Birk et al., 2013). A relevant feature of the assessment methods is that their 'state' response has to demonstrate a significant relationship with an anthropogenic pressure. Thus, the indicators used to measure ecological status fit clearly into the DPSIR scheme. Via the WFD (2000/60/EC) monitoring programmes established by the Member States, primary data on ecological status is available.

However, WFD implementation has not produced functional indicators explicitly linking state (S) and impact (I) according to the DPSIR scheme (Figure 1). Any (continued) potential for the delivery of ESS requires processes (e.g. reproduction of a fish population to allow for continued provision of fish as an ecosystem service). This represents a fundamental problem for analysing the relationship between status (which, at least in the WFD (2000/60/EC), is focussing on elements and not on processes) and services.

Ecosystem processes – such as primary production, ecosystem respiration, nutrient processing or decomposition of organic matter – are key features of the ecosystem that are framed by its elements and structure and determine, in part, its services. Functional indicators (e.g. leaf litter decomposition, ecosystem metabolism) have often been recommended to supplement ecological status assessment (e.g. Young et al. (2008), Sandin and Solimini (2009)), but rarely been implemented as the WFD explicitly focuses on ecosystem structure (e.g. composition and abundance of the biological communities (Birk et al., 2012)). But even functional indicators would be hard to link to services, as services are often delivered by a suite of processes and also human capital. What is a “good” or “bad” level of leaf litter decomposition? What service does it relate to? Also, many services only relate in a small way to ecological structure and processes. Human capital and social context are often as important – particularly in delivering provisioning and cultural services.

An appealing way of addressing ecosystem processes, albeit indirectly, is the use of functional traits, i.e., measurable properties of an organism (or habitat) such as body size, longevity, feeding guild (Culp et al., 2011). In this context, Bello et al. (2010) defined ‘trait-service clusters’ relating ESS to functional traits necessary for service provision. Relevant steps in their proposed workflow to link ecosystem properties and services include (i) identifying the relevant service, (ii) identifying the related traits and (iii) defining biological traits influencing the key ecosystem processes relevant for service provision. Lavorel and Garnier (2002) highlight the need to focus on “effect traits” that determine the effects of organisms on ecosystem functions. However, they speak about the trait-concept being the ‘holy grail’, addressing the high expectations largely unfulfilled in practice, due to this lack of knowledge.

Indicators used to measure ESS

ESS are measured by ‘performance indicators’, quantifying the degree of ecosystem service provision. Further subdivision of the conceptual framework as done by Schröter et al. (2014) offers the distinction between indicators of service capacity (i.e. the potential of the ecosystem to provide service) and service flow (i.e. the actual use of a service). This distinction enhances empirical clarity on the existence of actually used services versus the ecosystem potential to provide services. It can thus indicate whether the human use of the ecosystem is sustainable (in terms of flow not exceeding long-term capacity to provide these flows). In this regard, many authors applied simple ratios of service capacity and flow in their quantification studies of ecosystem service provision, measuring capacity and flow with different indicators (that operate at same units to allow for budgeting; e.g. Burkhard et al., 2012; Nedkov and Burkhard, 2012; Boithias et al., 2014; Schröter et al., 2014). In this contribution we pursue a biophysical perspective, thus focussing on indicators of service capacity and flow.

Currently, two major issues of service indication confound the sound application of the concept: the conceptual fuzziness in indicator selection (Layke et al., 2012) and the missing empirical linkage between ecosystem properties, functions and ESS (Keeler et al., 2012). In many studies putative service indicators are applied that in fact address driving factors (D), pressures (P) or states (S) according to the DPSIR scheme. Acknowledging this shortcoming in their analysis of existing service indicators, Layke et al. (2012) conclude: “While it may prove necessary to continue using proxies like ecosystem condition as indicators for some ESS, research to positively link the proxy indicator to the service should be undertaken”. As discussed earlier, concepts such as functional species or habitat traits hold promise for being able to establish these links.

Indicator goodness

A basic requirement for valid indicator selection is to conceive the linkage between indicator and *indicandum* more concretely (Müller and Burkhard, 2012). This requires the determination as to what extent the service indication represents the actual ecosystem service. We call this the indicator’s “level of proximity”. This level relates to the indicator type (e.g. whether it is a ‘true’ service indicator; or rather a ‘proxy indicator’ of the driving forces, pressures or states) and the type of data available to calculate the indicator (e.g. primary data from within study area, modelled based on data from within study area, data/evidence from outside the study area (Eigenbrod et al., 2010)).

A simple scheme for evaluating the indicator proximity is the distinction between ecosystem structure-related and ecosystem process-related services (Table 4). The former comprise services like *commercial fisheries*, *water supply* or *lake recreation*, measurable via state indicators such as biomass of edible fish, water quantity or bathing water quality. As these indicators are capable of directly measuring the service capacity of the ecosystem, they feature a high ‘level of proximity’. The indication of process-related services such as *self-purification* or *nutrient retention* requires functional indicators, at best measuring the relevant processes in the ecosystem. Modelling provides a viable alternative to infer process-related services, but model setup is elaborate and the outcomes often reveal high uncertainties (Hejzlar et al., 2009). Less elaborate process estimates are gained from extrapolating rule-of-thumb figures based on coherent conceptual models, representing a practical and fit-for-purpose service indication (e.g. Scholz et al. (2012)).

In conclusion, an inconsiderate use of state indicators for process-related ESS seems generally inappropriate. Ecosystem state and functioning are “intricately linked and describe different aspects of the same entity” (Young et al., 2008) – unravelling their relationship thus requires empirical investigation on the basis of individual case studies.

Table 4. Examples of ecosystem structure-related and ecosystem process-related services

Services related to ecosystem structure	Services related to ecosystem processes
• Food provision • Water supply • Biodiversity preservation	• Self-purification • Nutrient retention • Filtration of pollutants

• Recreation: bathing water quality	• Flood control • Erosion/sediment control

Research needs

The main research questions to be answered in DESSIN are as follows:

We propose to investigate into the links of ecological status and ESS on the basis of selected case studies derived from literature review and relevant projects (DESSIN, MARS, OpenNESS). These examples will help to verify if distinct relationships between provisioning, regulating and cultural services and ecosystem state can be assumed, as has been expected by Braat and de Groot (2012) or Kandziora et al. (2013). Quantifying relevant ESS in each of the DESSIN case studies on the basis of existing data from past and current ecosystem states will provide further experience in the application of indicators for ESS. Furthermore, we will apply process-oriented indicators from earlier studies (e.g. Scholz et al. (2012)) where applicable. Additionally, process-oriented indicators will be modified and adapted or new process-oriented indicators will be developed in order to perform sound investigations into the link between state and impact.

Determine indicators/proxies to measure changes in ESS

The challenge we face in WP11 of DESSIN is to illustrate the linkages between elements of freshwater ecosystems and changes in ESS that are relevant to the water sector. Proxy indicators need to be identified (and this can be either quantitative and/or qualitative). In this context and building from the findings of the previous sections, we would have some requirements that need to be met for the selection of proxy indicators:

- The selection of proxies will depend on the study area, its geographic extension and the availability of appropriate datasets.
- The method has to be mindful of valuation following quantification, and of possible aggregation of the value of multiple ESS.
- A need to find agreed criteria on data availability, policy relevance, ecological soundness, methodological soundness and the linkages with the selected ecosystem elements.
- The selected proxy indicators should be varied in scope (i.e. different in what they are measuring) to ensure that the complexity (i.e. biological diversity) of the studied ecosystems is covered without overlaps.
- A need to develop a detailed checklist for proxy indicator criteria (relevant for tasks 11.1.2 and 3 to inform task 11.1.1).

In terms of proxy identification, the criteria proposed can be:

Policy criteria: This describes the connection with policy targets (i.e. a 1x1 quantitative link) and policy implementation of appropriate datasets and information. The timing of policy implementation is an important aspect (e.g. in terms of availability of data and indicators). In particular, this is important in regard to the need to define targets for the distance to target calculation (expected changes).

Methodological criteria: This refers to the selected analytical framework. In other words, "does the proxy indicator support us in answering the questions we are asking"? This also means that proxies should be attained by following and improving internationally accepted standards, guidelines and good practices.

Data criteria: This refers to the need to ensure that data and metadata are presented in a clear and understandable manner as well as on an easily available and impartial basis. Thus, it is also important that products and services exist that ensure that data continues to be available at a required level of performance (i.e. from the mature case studies to the demo sites). Data quality and availability is achieved through redundancy, which involves information on where the data is stored and how it can be accessed.

3. Analytical framework to evaluate changes in ESS

The principal objective of DESSIN WP11 is to develop an analytical framework to evaluate and account for impacts that result from changes in ESS in the water sector. The main challenge in meeting this objective is to adequately link the relevant biophysical and social elements of the systems under study and, furthermore, to provide a transparent and robust approach applicable in practice. The DPSIR adaptive management scheme is helpful in disentangling biophysical and social elements in order to enhance conceptual clarity. An evaluation framework, however, needs to combine these elements to allow a holistic examination of the system under study.

A review of literature reveals multiple existing ESS assessment frameworks. The most accepted of which include the Millennium Ecosystem Assessment (MA) and The Economics and Ecosystems of Biodiversity (TEEB) which lay out methodological frameworks linking ESS to human well-being. However, these frameworks lack the ability to identify how changes in ecosystems will impact the provision of ESS. For the purpose of DESSIN's needs, it is necessary to look beyond the MA and TEEB frameworks to identify other opportunities which better identify and develop this link.

This chapter provides a brief review of a selection of existing frameworks, methodologies, and concepts which may be suitable for DESSIN. The advantages and disadvantages of each of the approaches are discussed. Please note: some of the figures in this chapter have been adapted from their original sources to comply with the agreed terminology for this report.

1. Ecosystem Service Profile (ESP) approach

Paetzold et al. (2010) present a conceptual framework for an ecosystem service assessment. The framework is based on the notion that the state of an ecosystem should be assessed in terms of its biophysical ability to provide ESS in relation to the societal use of those services. This approach, therefore, extends the linkage between ecosystem state and impact. This link between provision and use can be seen as a scheme in ecosystem service research, owing to its evident and illustrative character (Figure 3).

Figure 3. Scheme linking ecosystem integrity, ecosystem services and human well-being as provision and use sides in socio-ecological systems (adapted from Burkhard et al., 2012)

The framework allows for evaluations of individual ESS on the basis of ESS provision to ESS use ratios (referred to as the ratio of sustainable provision of an ESS to its expected level of provision, the latter also called demand, by the authors). This metric is useful for identifying provision versus use balances for individual services across water bodies, sub-catchments or entire water basins. Nedkov and Burkhard (2012), for instance, analyse the ESS provision to ESS use budget (named water supply:demand ratios by the authors) of a Bulgarian sub-catchment to map flood regulating ESS. Boithias et al. (2014) assess water ratios (labelled as supply:demand ratios) in a Mediterranean basin under different global change scenarios and mitigation alternatives. Schröter et al. (2014) account for the provision and use (referred to as capacity and flow of ESS by the authors) of various terrestrial ESS in the Norwegian countryside.

Each individual service is defined by the ratio (R) of its actual provision (P) to the expected or realised level of service use (U). Figure 4 depicts an ESP for a hypothetical example of selected ESS. The upper panels show levels of provision and use of ESS. Note that for different ESS such levels can be expressed in different units, whereas for a single service provision and use are measured in the same unit. The lower panels show the respective ESPs expressed by the ratio of provision to use of the individual services. A ratio of P:U < 1 means that use is exceeding provision, P:U > 1 means that provision is in excess of societal uses, and P:U = 1, where P equals U, indicators sustainable usage of a service. Higher ratios (R > 1) represent further buffering capacity against potential fluctuations in both the provision and use of an ecosystem service.

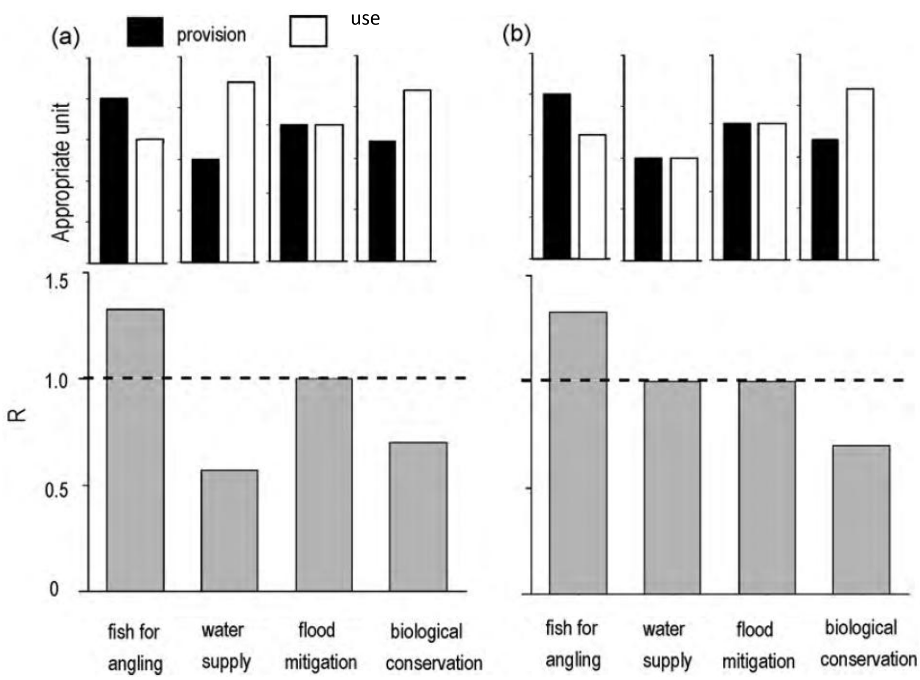

Figure 4. Ecosystem Service Profile (adapted from Paetzold et al., 2010). a) The upper panels show provision (P, black bars) and use (U, white bars) quantified for each of the four exemplary ESS. The lower graph depicts P:U ratios (R) determined for each of the services. The dotted line indicates where provision equals use. b) A scenario of an improved ESP resulting from the reduction in use for one of the services, while provision is kept constant

To establish an ESP for a set of selected ESS, Paetzold et al. (2010) generated a stepwise methodology. Figure 5 depicts the major steps in the development of an ESP. These are:

(1) Identify all ESS of potential relevance in your case-study.
(2) Communicate the services to link societal interests to them on the basis of stakeholder exchange. For a single case-study, ESPs can be developed for several stakeholder groups in order to include different viewpoints which can, subsequently, be compared.
(3) Select the most relevant and representative services, considering stakeholder opinion and existing legislation.
(4) Define pair-wise sets of suitable service indicators, representing (i) service provision and (ii) service use. Provision indicators need to be assessed in the same units as the respective use indicators.
(5) Quantify the level of provision and use for each ecosystem service.
(6) Derive the ESP by calculating the provision-to-use ratio for all ESS.

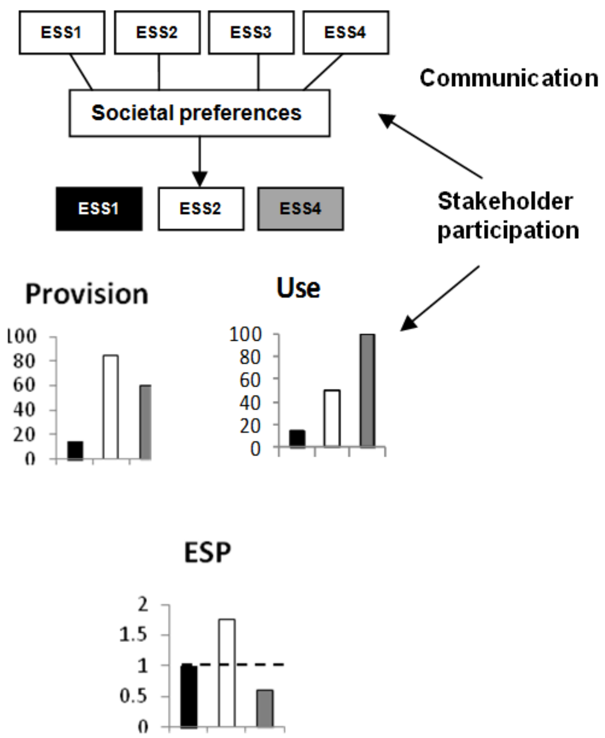

Figure 5. Major steps in the development of an Ecosystem Service Profile (ESP) (adapted from Paetzold et al., 2010)

The ESP concept includes both social and biophysical aspects in ecosystem service assessments. This implies that ecosystem management cannot simply be based on increasing service provision but must also look at options to manage expectations and uses. Furthermore, the framework:

- accommodates the variability of service provision and use, depending on context and time; Paetzold et al. (2010) suggest to first defining the spatial and temporal context for the

assessment of ESS but then also to consider the spatial linkages among service provisions and uses within and outside the defined area.
- allows for service trade-off analysis;
- recognises the notion of sustainability; and
- incorporates legislative requirements (e.g. coming from the WFD (2000/60/EC)) – not as exclusive goals but as single relevant components among others.

It even provides a tool for assessing changes in the ESP under varying degrees of anthropogenic pressure. It can, for example, be applied to compare environmental systems at various points in time (e.g. past versus present versus future scenarios) or under various management scenarios. The benefits of using this approach include the direct linkage of management practices with provision and use of ESS. In particular, this approach can facilitate the efficiency valuation of innovative technologies for the water sector without requiring extensive databases and marked baselines. If the ESP for a selected set of water services is evaluated before and after the implementation of an innovative water technology, economists can potentially extract the marginal changes in the provision of water ESS and conduct economic efficiency valuations from these two states.

2. Water Quality and Well-being

In an effort to make water quality assessments more meaningful to the public and interested stakeholders, Keeler et al. (2012) have developed a generalisable framework for the assessment and valuation of water quality services (Figure 6). This generalisable framework integrates biophysical and economic models, bases value estimates on marginal changes in service provision and accounts for multiple sources of value without double-counting. Though this proposed framework is specific to water quality, there are possibilities to extend the framework to include changes in water quantity for the purposes of DESSIN.

This framework consists of five steps as described below (Keeler et al., 2012):

1. *Identify actions and beneficiaries of interest*: identifying the beneficiaries of interest and then working backward to determine the appropriate biophysical parameters that have the greatest potential to affect those groups provides focus for research efforts and can ensure that subsequent work captures the most important drivers and ecosystem service consequences. If water quality information is already available, the framework can be used to identify all the potential services affected by a change in a given nutrient or pollutant concentration.

2. *Identify shared inputs/outputs of biophysical and economic models*: identifying parameters which need to be included in a set of integrated biophysical and economic models. These parameters describe aspects of water quality that can be measured or modelled in biophysical assessments and directly affect human well-being. The choice of parameters depends on biophysical understanding, links to human well-being and data availability.

3. *Select appropriate biophysical models*: identifying an appropriate biophysical model to capture the effects of an action on a parameter at a defined endpoint. Comprehensive

valuation of water quality may require different biophysical models for each water quality constituent.

4. *Select appropriate economic models*: identifying an appropriate economic model to link valued parameters at particular endpoints with economic models that measure the value of these parameters to specific beneficiaries. Economic models can be used to compare the well-being of people before and after a change in water quality. Economic models should measure change in value in terms of a common monetary metric.
5. *Consider existing models and data sources*: in some cases, existing work is sufficient to translate biophysical outputs to change in service provision and value; however, few generalisable models linking actions to changes in value exist for water-quality related services. In many instances, researchers will have to collect new data in their region of interest or make assumptions about how to adapt existing models developed in other contexts (see Chapter 5 'Benefits Transfer').

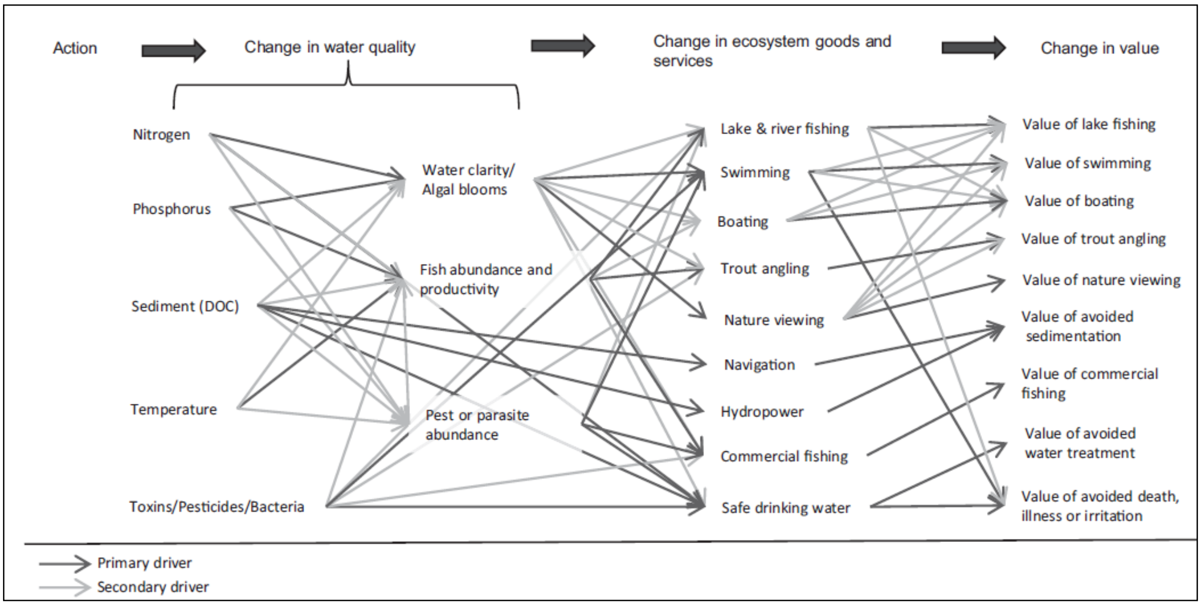

Figure 6. Relationships between water quality change, multiple ecosystem services and associated changes in values. Connections between columns are classified as primary or secondary, according to expert opinion (Keeler et al., 2012)

Benefits of using the Water Quality and Well-being framework proposed by Keeler et al. (2012) mainly focus on the integrative aspects of biophysical and economic research and models. Theoretically, this type of assessment could be applied to multiple scales, though the authors do not explicitly state such a claim. However, they do claim that this framework is comprehensive, avoids double-counting of costs or benefits through explicit delineation of water quality intermediate[23] and final[24] ESS, and is sensitive to alternative land use or management decisions

[23] Also called supporting services which are underlying ecological structures, processes and functions, e.g. nutrient cycling.

(Keeler et al., 2012). Lastly, this framework links actions to measured/modelled marginal changes in water quality and their subsequent effect on changes in the value of ecosystem goods and services (Keeler et al., 2012). This has important ramifications for decision-making processes which can then better account the marginal benefits of additional ESS provision.

The drawbacks associated with this framework mainly revolve around time and data requirements. The authors openly state that the valuation part of their framework is time-consuming and requires carefully consideration of assumptions and uncertainties in the models as well as the framework pathway (Keeler et al., 2012). This concern is linked to the heavy reliance upon both the quantity and quality of data available to use as inputs for both the biophysical and economic models. Without the appropriate information at hand, the uncertainties associated with the modelled and measured results increase. The financial cost of this data collection and the associated measuring/modelling aspects are not clearly stated. Lastly, this framework is also very explicitly geared towards water quality which may marginalise other water-related ESS.

3. TESSA

Toolkit for Ecosystem Service Site-based Assessment (TESSA) is a site-scale guide for local non-specialists to select relatively accessible methods for identifying which ESS may be important at a site and to evaluate the magnitude of benefits that people obtain from them currently, compared with those expected under alternative land-uses. Information at a site-scale is valuable for determining whether there are utilitarian and intrinsic arguments in support of conserving particular areas, and for informing decision-makers whether conserving (rather than converting), or restoring, a site has broader benefits for society (Peh et al., 2013). Consequently, TESSA provides a net benefits framework through applying a set of appropriate methods for two alternative states of a site.

Peh et al. (2013) argue that methods for quantifying services need to produce data relevant to decisions affecting that site, should be practical and affordable (in terms of expertise, equipment and time) and should provide results in an accessible form to actors such as policy-makers, planners and land managers. To this end, TESSA uses a reduced set of ecosystem service categories which was selected based on importance and measurement tractability: global climate-regulating services, water-related services, harvested wild goods, cultivated goods, and nature-based recreation.

[24] „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." (Haines-Young and Potschin, 2013) (CICES)

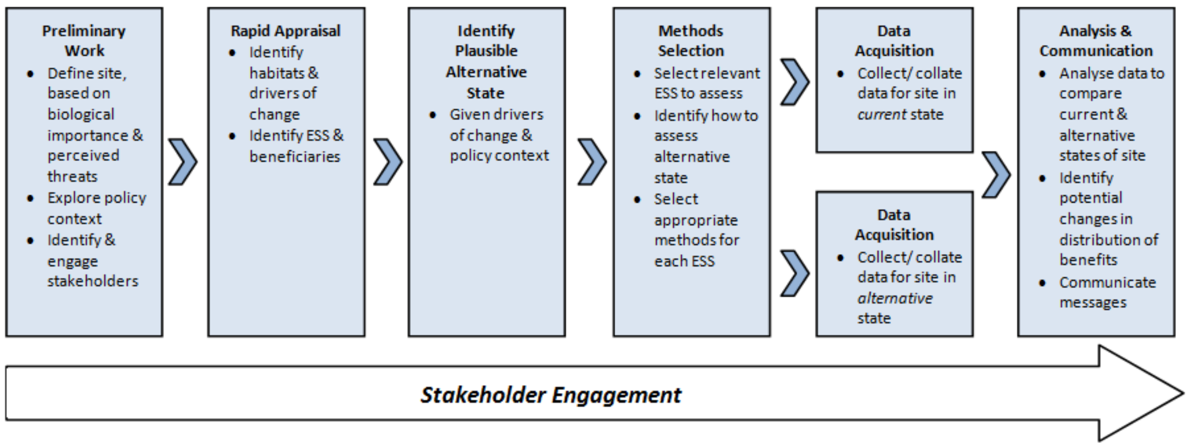

Figure 7. TESSA methodological framework (adapted from Peh et al., 2013)

The TESSA methodological framework is outlined in Figure 7 which begins with defining the site of interest based on its biological importance and perceived threats, exploring the local policy and governance context and identifying stakeholders. This step is followed by a rapid appraisal that identifies the most important habitats, drivers of land-use change and the services provided by the site. Further assessment then focuses on services that are (i) significant in either biophysical, social or economic terms; (ii) sensitive to potential drivers of change; and (iii) measurable with limited capacity and resources. The third step then identifies the most plausible alternative state of the site (Peh et al., 2013).

Once these initial steps have been taken, TESSA guides the user through decision trees to appropriate methods for each service. These include collecting primary data through field surveys, key informant interviews, and household questionnaires, using existing databases and studies, and employing numerical models. The chosen method will depend on the availability of time, resources, expertise and on the extent to which useful data have already been collected.

Additionally, TESSA includes guidance on how to assess the distribution of benefits between stakeholders both according to spatial scale and among different socio-economic groups, as well as how to communicate findings (e.g. through bar charts or rose plots (see Figure 8) (Peh et al., 2013). Use of this feature and of the TESSA toolkit has been incorporated into 10 case studies (Peh et al., 2013) and a recent study on benefits of community forests in Nepal (Birch et al., 2014).

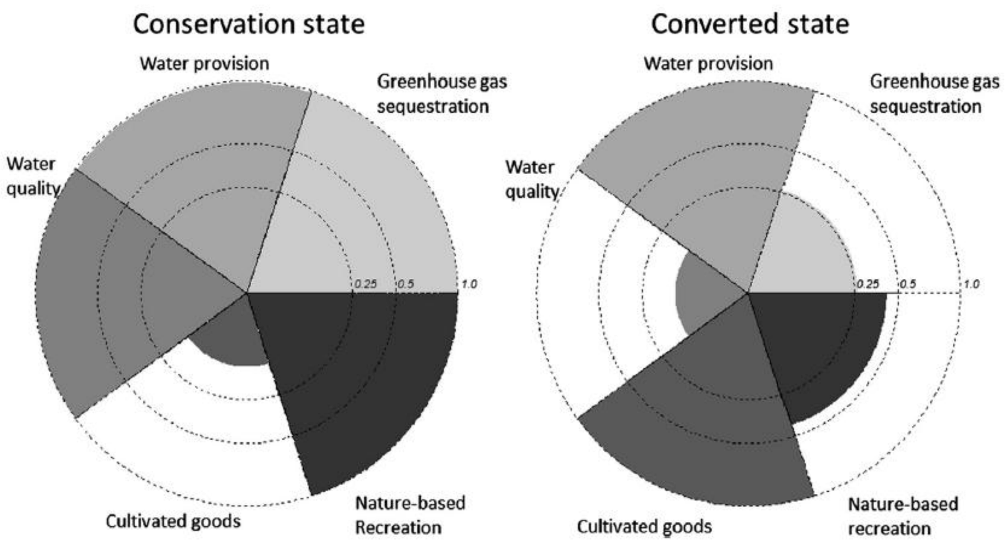

Figure 8. TESSA example of how to communicate findings to decision-makers and stakeholders (Peh et al., 2013)

According to Peh et al. (2013), TESSA's main strength is that it empowers local users and non-specialists to engage in ESS assessments through the use of methods that are flexible, adaptable to time and capacity constraints and are designed by experts in each service. Consequently, TESSA users gain valuable information regarding the alternative states for a site and their associated benefits and drawbacks. TESSA's inclusion of social benefits, distribution of ESS goods and services, as well as recommendations for communication of results is another strength of this toolkit; one which attempts to bridge the gap between science, policy and the public. The collected field data also has the potential to be incorporated into regular monitoring programmes (Peh et al., 2013). Financially, TESSA has demonstrated low application costs at four pilot sites: 13-49 days (median 39) of personnel time in the field, £1,000– £6,000 (median £4,200) for equipment costs (Peh et al., 2013). This is important for areas where ESS assessments are obstructed or impeded due to financial constraints and may allow for their wider implementation.

Weaknesses of this toolkit are clearly explained by Peh et al. (2013) and include the limited selection of ESS included in the toolkit, the exclusion of sustainability and resilience concepts, the lack of temporal variation of ESS provision, the exclusion of non-linearities and tipping points within ecosystem dynamics and their effects on ESS provision and the lack of climate change projections due to the focus of threats on shorter time scales. An additional drawback is the scale this toolkit is designed for (site-scale), which limits attempts at upscaling, though this is understandable due to the toolkit's fieldwork and stakeholder involvement aspects. Lastly, though the current version of TESSA enables users to derive monetary values for some ESS, generating economic values for water-related ESS has proved more difficult (Peh et al., 2013) and, thus, poses a significant drawback for the purposes of DESSIN's work.

4. RUBICODE

Components of the Rationalising Biodiversity Conservation in Dynamic Ecosystems (RUBICODE) project have been applied to freshwater and terrestrial ecosystems in Europe in order to provide frameworks to rationalise biodiversity conservation strategies (Harrison, 2010). RUBICODE builds upon the work of the MA (2005) to review and advance methodologies for assessing the state of Europe's ESS.

RUBICODE's Framework for Ecosystem Service Provision (FESP) was developed to assess the impacts of direct and indirect drivers on ESS and to identify the mechanisms of mitigation or adaptation derived from policy or management responses (Rounsevell et al., 2010). This framework allows comparison across competing ESS, highlighting the conflicts and trade-offs between not only multiple ESS but also multiple service beneficiaries (Harrison, 2010). FESP is based on the DPSIR scheme with introduced elements to the ecosystem service approach (Rounsevell et al., 2010) and is, therefore, in line with DESSIN's needs. This FESP framework is graphically depicted in Figure 9 and illustrates how drivers of environmental change impact ESS and how policy and management responses are derived from the valuation of these impacts.

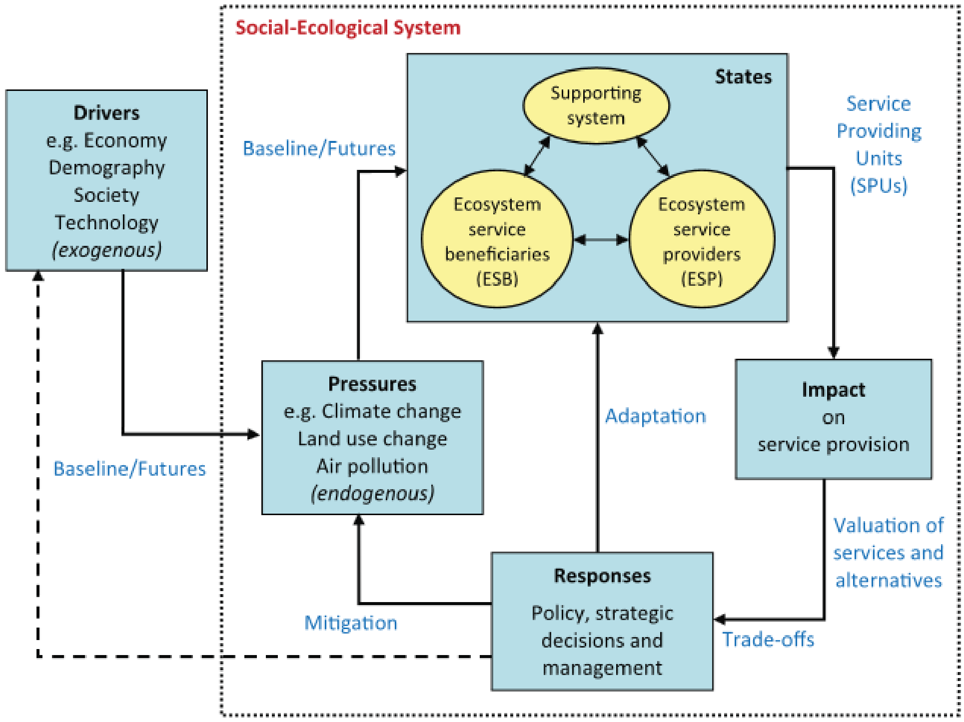

Figure 9. FESP adaptation of DPSIR scheme (Harrison, 2011)

RUBICODE was initially designed to aid decision-making for biodiversity conservation by taking into account ecosystem dynamics and land and other resource constraints. The FESP framework developed in RUBICODE has already been applied to freshwater ecosystems for water-related ESS such as water purification, water cycling and food provision (Harrison, 2010; Rounsevell et al., 2010; Lavorel et al., 2009.

The methodology to identify and quantify ESS within the FESP is depicted in Figure 10, broadly following three steps: (i) identify the beneficiaries of the ESS and the biological organisms that provide it; (ii) quantify the use of the ESS and compare this with the provision of the ESS; and (iii) determine the value of the ESS, identify alternatives to ESS provision (e.g. human-based actions) and assess policy and management implications (Harrison, 2011).

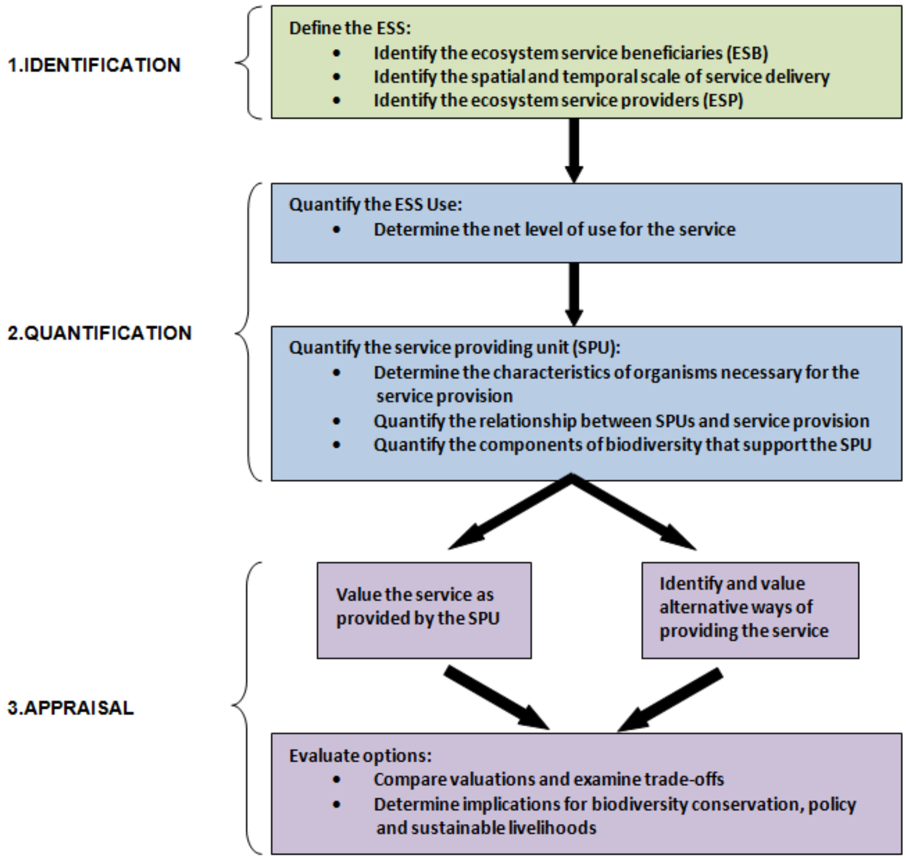

Figure 10. RUBICODE guidelines for identification and quantification of ESS (adapted from Harrison, 2011)

Benefits of using RUBICODE are mainly that it makes the comparison across competing services clear and accessible, as well as making the trade-offs and conflicts between multiple ESS and between multiple service beneficiaries explicit. The FESP can also identify the mechanisms of mitigation or adaptation to environmental change issues through the effect of response strategies on specific pressure or state variables (Rounsevell et al., 2010). With respect to DESSIN, the RUBICODE framework has an advantage in that it already uses an adapted DPSIR scheme to account for the ESA. This is in line with the work proposed within DESSIN and has the potential for a relatively easy integration. The fact that this framework has already been applied to freshwater ecosystems is another advantage.

The FESP framework, however, has its limitations. As stated by Rounsevell et al. (2010), the framework needs more comprehensive testing against a wider range of practicable examples and does not describe in detail the human or biophysical processes that compose complex social-ecological systems. It also doesn't explicitly state where stakeholder participation would be incorporated, or even if there is an aspect of stakeholder participation.

5. Blueprints as structural templates for ESS assessment studies

Seppelt et al. (2012) propose to use blueprints for a consistent and complete reporting of ESS assessment studies. The suggested template consists of 5 sections, being 1) *Purpose and design*, 2) *Problemscape and concept*, 3) *Analysis, assessment, test*, 4) *Recommendations and results*, and 5) *Monitoring*. Table 5 depicts an example from Seppelt et al. (2012).

Such blueprints can be of major advantage for a consistent documentation of ESS assessment studies in order to allow comparability between studies. With this standardised form, the studies can additionally feed into scientific databases of ESS investigations. This would enhance the available knowledge base for comparisons and meta-studies, the exchange of experience and reproducibility.

Table 5. Example for a blueprint for ESS assessment studies (adapted from Seppelt et al. 2012)

Purpose and Design	• Scope of the study • Project goals • Main threats • Targets • Stakeholders • Team of scientists
Problemscape and Concept	• System description • Ecosystem services • Landscape • Policy measures • Expectations/challenges • Storylines of potential futures
Analysis, Assessment, Test	• Indicators (derivation based on DPSIR model) • Ecosystem service indicator calculation • Models • Scenario quantification • Valuation and test
Recommendation and Results	• Trade-off analysis and off-site effects • Recommendations
Monitoring	• *Possibly establishment of monitoring at the end of the project*

Discussion

The aim in DESSIN is to promote innovations in the water sector with the help of the ESA. Towards this aim, the ESA can translate effects on the ecosystem resulting from the implementation of technical innovations into economic value in terms of ESS. In the end, the possibility to compare investment costs for the implementation of technological measures in relation to its benefit in terms of ESS opens up new ways for promoting these technologies. Ideally, feasibility and cost-benefit studies should always be accomplished by ESS assessment.

Additional considerations

Proper application of any analytical framework requires appropriate indicators regarding both provision and use. This relates to the flaws in the current indication of ESS, addressed in Chapter 2: "Besides a common conceptual fuzziness with putative service indicators in fact addressing driving factors, pressures or states, empirical linkage between ecosystem properties, functions and ESS is often lacking." The selection of appropriate indicators of ecosystem service use (and the quantification of said use) is also challenging and implies intense stakeholder involvement, if values cannot be derived directly from policies and legislation.

Scale is another aspect that requires further attention in the implementation of the ESS assessment framework. Different ESS are provided and used at different spatial and temporal scales. Careful consideration of spatial and temporal issues must be incorporated into the DESSIN framework to ensure consistency and comparability of results (see Chapter 4).

Furthermore, trade-offs between services need to be identified, considered and discussed, and double-counting of ESS should be considered and avoided. Finally, the uncertainties associated with the application of the selected ESS need to be addressed.

The selection of water-related ESS, indicators, choice of economic and biophysical models, means of measurement and choice of valuation methods (see Chapter 5) are all based on value judgments, which can potentially influence the results of ESS assessments.

4. Spatial and temporal issues in relation to ESS

The benefits of ESS may be removed in space from the ecosystem providing them. The spatial distribution can act on local, regional as well as global scales. Furthermore, the ESS may change over time both seasonally and in a more long-term perspective. For some changes in pressures, there may be a time lag before changes in pressure and in ecosystem status manifest themselves in changed ESS. Spatial and temporal issues can arise when some ESS depend on interactions between ecosystems or between different temporal stages. Furthermore, the responses of ecosystem functions and thereby ESS to changes in pressures are not necessarily linear (Koch et al., 2009).

For evaluation of these temporal and spatial issues, it is important to build on specific and often local knowledge about ecosystem interaction and function. Therefore, it is not possible to provide generic temporal and spatial guidelines for measuring changes in ESS. In addition, addressing spatial and temporal variation in ESS often requires either high resolution monitoring data or a modelling approach that can describe these variations. When working with ESS, it is therefore important to use an approach and method that are flexible and make it possible to expand and modify the assessment both in a spatial and temporal context.

As data resolution often is a limiting factor, modelling support for identification and quantification of ecosystem functions and services provided may be attractive. In addition to facilitating better understanding of temporal and spatial issues, models can often clarify new and important relationships between services provided by different ecosystems. However, such models should capture a high degree of ecosystem functionality and also need to be ecosystem specific. The result is that building and calibrating these models can be resource intensive.

The challenge of defining the spatial scale in relation to definitions of ESS is highlighted by Potschin and Haines-Young (2011), who conclude that it is important to work beyond the traditional definitions based on natural science-based units such as habitats and catchments. They further recommend working with space-to-space relationships. They conclude that there may not be one single appropriate scale for measuring a given ecosystem service and therefore a cross- and multi-scale approach may be required, as different scales may be relevant for understanding issues in play at different places. Even within the same area, different temporal and spatial resolutions can be required for identification of different ESS.

Although it is not possible to provide generic guidelines, it is possible to recommend an outline of some issues. Some examples are provided below:

- ESAWADI (2013) gave warning to challenges with scaling and mentioned that the choice of scale can change the issues to be included in an assessment as well as the choice of stakeholders to be involved. The upstream-downstream issue and the interactions have been identified as such a challenge. A creek ecosystem providing spawning and rearing localities for trout and salmon is an example of an upstream-downstream issue. This is because the localities are essential for fish production and fishery ESS provided by marine ecosystems that can be far away from the creek ecosystem.

- Amigues J.P. and Chevassus-au-Louis B. (2011) stressed that the unit to which the services are related must be defined carefully. The authors have invented a unit named “hydrosystems”, which goes beyond the conventional notion of ecosystems – both with respect to spatial and to time resolution. The defined “hydrosystems” include physical, biological and socio-economic dimensions of an aquatic environment and recommend a step-wise approach for identification of the appropriate resolution.
- (Cardoso et al., 2013) mention in a working paper in relation to the MAES Pilot 4 project that the identification and mapping of ecosystems depend largely on the availability of land cover/land use datasets at various spatial resolutions. The best available dataset at the EU level is the Corine Land Cover. Corine data is not always sufficient for providing a sound picture. Kandziora et al. (2013) show how land use and land cover data could be used to describe spatial and temporal variation in provisioning ESS and that different results can be achieved using different data sources.

Mapping of spatial data and addressing differences in spatial ESS should not be limited to land cover data only. Ecosystem diversity and function may not be represented adequately using land cover data and may need additional data such as elevation and climate statistics, more detailed biomonitoring data and information about cultural and sociological information mapping of ESS at appropriate scales.

The EU Water Framework Directive (WFD) (2000/60/EC) and the Floods Directive (FD) (2007/60/EC) are generally concerned with water management at the river basin level, and the WFD ecological status descriptions are in some cases operational at smaller scales. When addressing the spatial issue, these scales should be considered, but it may be useful to look for services beyond these traditional environmental management scales. The FD gives guidance for different units of management.

Some ESS are appropriately defined at small scales and others at larger units such as the river basin. In some cases, it is necessary to consider different scales simultaneously to be able to identify, describe and understand different ESS. For some biodiversity issues (e.g. for some floral protection), a smaller scale is appropriate. For others (e.g. protection of migrant birds), larger scales should be considered. For cultural and provisioning services, a scaling beyond the traditional understanding of the ecosystem boundary may be required.

Examples of ESS that may require consideration of larger spatial scales include: greenhouse gas/carbon sequestration in floodplains, wetlands, lakes and the marine environment; aesthetics of landscapes with semi-natural/artificial aquatic elements; fish stock recruitment; nutrient filtration and immobilisation of pollutants; ESS provision from forestry; and cultural, recreation and tourist benefit services.

In relation to temporal resolution, understanding the dynamics of ecosystems and changes in dynamics are important factors. This becomes obvious when considering hydromorphological issues and related services. Flooding, for example, is by nature a temporal phenomenon. Delivery of natural flood storage capacity is an example of a service that must be provided with consideration for both time and space. However, flooding has both positive and negative impacts, and allowing controlled flooding can conserve biodiversity in wetlands (Amigues and Chevassus-au-Louis, 2011). In some rivers, preserving natural hydrological variation is a prerequisite for ensuring migration of species throughout the catchment area. It is not only the amount of water (discharge) that is important, but also the timing of the hydrograph may be important (i.e. the necessary storage capacity must be available at the right time).

The concept of environmental flows with different requirements in different seasons may assist in relation to temporal resolution. The EU "Blueprint to Safeguard Europe's Water Resources" (European Commission, 2012) also mentioned environmental flows as an element to be included in future WFD management guidelines.

In addition to seasonal considerations, more long-term changes may also change ESS provided by a specific area or ecosystem. An example of a long-term change is the well-known response occurring in lakes after pollution reduction (Jeppesen et al., 2005). Due to an accumulated sediment pool of pollutants (especially phosphorus), the lake ESS of nutrient retention can take decades to change after a pollution reduction (Søndergaard et al., 2003).

The provision of ESS relevant to fisheries, recreation and tourism may not show changes until a long time after measures for environmental improvement have been implemented. An example is the restoration of the River Skjern[25] in western Denmark, which led to changes in nutrient retention capacity, fish production, biodiversity protection and opportunities for recreation and tourism. The full effects of the restoration were first observed more than a decade after project implementation.

Social-economic and cultural changes can also result in changes to ESS provided and demanded. These changes can include the range of, as well as the quantity of, ESS and benefits provided. This may be especially relevant for provisioning ESS. Examples include social-economic changes that may have significant consequences for the demand for tourism-related ESS in an area, or that result in changes to demand for natural resources such as timber and water. Such impacts will most often be seen outside traditional, natural science-based ecosystem boundaries and, in the case of changes brought about by actions such as adaption and awareness-raising campaigns, may not be visible for some time after the initiation of these actions.

[25] For more info visit: http://www.skjernaa.info

5. Economic valuation of changes in ESS

The concepts of 'value' and 'valuation' have many meanings and a long history in several disciplines (Farber et al., 2002). Ecological valuation is generally based on bio-physical accounting most often with total neglect of human needs and/or wants. Contrarily, economic valuation is based upon consumer preferences and therefore takes human needs into account (Spangenberg and Settele, 2010). In this context, the value which users derive from an ecosystem service is depicted in the total economic value. The total economic value placed on environmental assets can be disaggregated into use and non-use values:

- ***Use Values*** arise from the actual and/or planned use of the service by an individual. Use values can be *Direct Use Values*, such as when an individual makes actual use of the environmental asset improved; or *Indirect Use Values*, taking the form of ancillary benefits derived from ecosystem functions gained or recovered.
- ***Non-use Values*** arise independently of any actual or prospective use by the individual. These are usually categorised as *Existence Values*, which arise from knowledge that the service exists and will continue to exist; and *Bequest* or *Option Values*, which measure individuals' preferences to ensure that the service will be available for their own use in the future and that future generations will also have access to the service.

All of these values can be estimated by using market-based valuation methods or by analysing revealed and/or stated preferences of users. The variety of existing valuation methods is described in this chapter.

The purpose of this chapter is to review ways to account for values associated with societal preferences for changes in water ESS provision. An introduction and a discussion on the state-of-the-art for each economic valuation methods is given. Under each of these specific valuation tools, there is a discussion of their applicability in the water sector. Finally, this discussion is also used to come up with a classification of valuation methods that agglutinates recent developments/views in the valuation literature (see Figure 11 below).

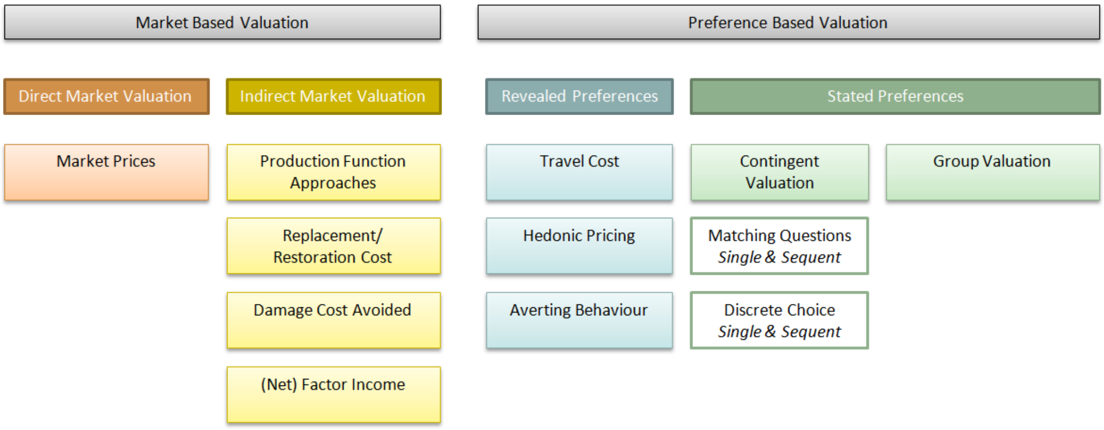

Figure 11. Classification of valuation methods. Source: Own elaboration based on the most consistent elements of the typologies described below

Market-based environmental valuation methods

Direct market valuation

Direct market valuation is only applicable where a market exists for the ESS and data is readily available. In this respect, market prices can be seen as valuations directly obtained from what people or firms must be willing to pay for a service or good (Farber et al. 2006). Direct market valuation acts as a means of assessing the value of ESS in monetary terms, i.e., the exchange of ESS for money within a market (De Groot et al. 2002; Spangenberg and Settele 2010; Farber et al. 2006). However, caution must used when considering this valuation method due to price distortions from market-based interventions (e.g. taxes, price ceilings or floors, subsidies) as well as externalities which do not include the true social cost or benefit of a service in the market price (Turner et al., 2010).

In relation to water ESS valuation, market-based methods may be difficult to implement. One reason is that market prices are not directly established by society's "willingness to pay" in the drinking water market (essential service), as it is considered a natural monopoly with high capital requirements and is typically regulated. Another reason is that water provision might be nationally subsidised, as done within many member states of the EU. For these reasons, any direct market-based economic valuation done for water services can result in inappropriate estimations for water-related ESS.

Indirect market valuation

When no explicit markets for ESS exist, indirect market valuation methods are used as indirect means of assessing values (de Groot et al., 2002; Spangenberg and Settele, 2010; Turner et al. 2010; Birol et al. 2006). Consequently, market prices act as an accounting procedure that can also be extended to other non-market ecosystem service benefits by observing how changes in their provision affect the prices or quantities of other related marketed goods (Turner et al. 2010).

Production function approaches

Production can be influenced by the environment in various ways, e.g., by changing the productivity of inputs, by altering the quality of the output or by reducing the effective supply of inputs. These effects can be modelled by treating the environment as an input in the production function (Bockstael and McConnell, 2007). Different methods exist where the physical changes in output due to environmental changes and damages are measured through the usage of market prices or costs to value these impacts, e.g., dose-response, change in productivity and damage function models (Hanley et al., 2009).

Production function approaches generally estimate the contribution an ESS makes to the production of some marketed/marketable service (Chee, 2004; Farber et al., 2006) or, in other words, isolate the effect of ESS as inputs to a production process (Bateman et al., 2011).

With respect to water ESS, production function approaches can be useful to estimate a partial value of a water ecosystem when there is a clear link between a water ecosystem and the production of an economically valuable commodity. The approach cannot be used to estimate non-use values. According to Bateman et al. (2011) examples of water related ESS valued with production function approaches exist for: supporting aquaculture, ground water recharge and drainage and natural irrigation. For example, estimations can be made from the reduction in agricultural or business output resulting from a reduced volume or quality of in-stream (aquifer) or off-stream (reservoirs) water (WBCSD, 2013).

The existence of market prices for water-produced commodities (e.g. commercially harvested fish) makes production-based valuation of use values for water ecosystems less controversial than most non-market methods. However, there remain a number of difficulties, especially in valuing urban water cycles with the production function approach. Reasons for this include spatial application bias (most studies applied in rural settings), limited valuation of water ESS (e.g. focus on increased fishery productivity), estimation influence from property rights and regulation, and difficulties establishing a clear quantitative link between water ecosystems and productivity due to influences from natural variation (Boyer and Polasky 2004).

Replacement cost, Restoration cost

The idea behind the replacement/restoration cost method (RCM) is that services could be replaced with human-made systems (de Groot et al., 2002; Farber et al., 2002). The RCM estimates the value of a change in a non-market natural system service by evaluating the cost of replacing the lost or reduced service with a manmade substitute service or by evaluating the cost of an ecosystem restoration (Chee, 2004; Farber et al., 2006; Turner et al., 2010; Bockstael and McConnell, 2007). This method cannot estimate non-use values.

According to Spangenberg and Settele (2010) the RCM can only be used as an approach, if certain conditions are satisfied. These include (1) the replacement system must provide functions/services that are qualitatively and quantitatively equivalent to the original ESS; (2) the replacement option must be the least cost option of all possible replacement options so as to avoid overestimation of the replaced ESS; and (3) the aggregated willingness to pay for the replacement must exceed the cost of the replacement in face of the loss of the original ecosystem functions, so as to avoid welfare loss.

All these points raise issues related to the real degree of substitutability among alternative projects. In addition, alternative artificial solution investment must ensure that adequate maintenance costs are included for a long enough period of time.

Within the water sector, RCM has been used to value disturbance regulation, water regulation, water supply and waste treatment ESS (de Groot et al., 2002). The replacement service valued typically focuses on a single ESS (e.g. water purification) capturing only a part of the value rather than the complete range of values associated with a water ecosystem. In this respect, human-made ESS replacements are rarely successful in substituting all of the services generated from the original ecosystem. RCM is particularly applicable where there is a standard that must be met, such as certain level of water quality.

Damage Cost Avoided, Avoidance Cost

The idea behind this approach is that services allow society to avoid costs that would have been incurred in the absence of those services (de Groot et al., 2002; Farber et al., 2002). A service is valued on the basis of costs avoided by not allowing ESS to degrade (Bateman et al., 2011).

Avoided costs can be used to evaluate the benefits of resource alternatives on the supply-side, including leak-detection and repair programmes, water purchases from alternative suppliers and source-of-supply or treatment options for complying with drinking water standards (Beecher, 2011). Other water-related ESS valuations include the avoided costs of dredging and avoided health costs of water or seafood contamination through value of a statistical life (VSL) estimates (Griffiths et al., 2012). The use of this valuation method is often found in welfare economic studies.

(Net) Factor Income

The (net) factor income approach is generally described as the enhancement of incomes a service provides (Farber et al., 2002; de Groot et al., 2002). The net factor income approach estimates changes in producer surplus by subtracting the costs of other inputs in production from total revenue, and ascribes the remaining surplus as the value of the environmental input (Brander et al., 2006; Birol et al., 2006).

Examples of this ESS valuation method in the water sector include water quality improvements which increase commercial fisheries catch (and thereby the incomes of fishermen), improvements in agricultural productivity and decreased costs of purifying municipal drinking water; as well as wetland ESS (de Groot et al., 2002; Birol et al., 2006; Brander et al., 2006).

Preference-based environmental valuation methods

The vast majority of ESS have no market price (Heal, 2000; Naidoo and Ricketts, 2006; Daily et al., 2000; Turner et al., 2003; as cited in Cowling et al., 2008), as neither directly nor indirectly real or hypothetical market prices can be determined (Spangenberg and Settele, 2010). In this case, price and value calculations are derived either from revealed preferences or from stated preferences (Spangenberg and Settele, 2010). Ultimately, data is collected that provides information from which one may infer social preferences (Carson and Louviere, 2011).

Revealed preference methods

Revealed preference methods (RP) are based on indirect calculations, deriving value figures from the effects of behavioural change associated with a service or the lack of a service (Spangenberg and Settele, 2010).

Averting behaviour

The averting behaviour approach can be defined as the examination of expenditures to avoid effects of environmental damage (Bateman et al., 2011). The method is based on the household production function theory of consumer behaviour (Birol et al., 2006), where marketed goods can act as substitutes for environmental quality or goods in certain circumstances. When a decline in environmental quality occurs, expenditures can be made to mitigate the effects and protect the household from perceived welfare reductions (Pearce and Howarth, 2001). This is largely limited to services related to properties, assets and economic activities, and is therefore limited to measuring use values. Averting expenditures obtained provide a lower bound estimate of the total costs imposed (Turner et al. 2010). The divergence between the averting expenditures and the total costs of environmental degradation arises as many consequences cannot be avoided (Courant and Porter, 1981).

Concerns regarding the use of this method focus on its ability to accurately measure willingness to pay. Courant and Porter (1981) argue that in general, the averting behaviour method is not a good measure of willingness to pay, with issues concerning the real degree of substitutability among alternative choices. The best case scenario would be that the goods are prefect substitutes or show a very high degree of substitutability (Turner et al., 2010). More difficulties arise when joint products are used as substitutes, as the value estimates have to be disentangled (Turner et al., 2010). The households should also not obtain direct utility from the averting behaviour (Committee on Valuing Ground Water, 1997).

For water related ESS, the averting behaviour method is applicable to water purification (Turner et al., 2010). However, the Committee on Valuing Ground Water (1997) points out that in most cases, information from averting behaviour studies will need to be coupled with and in some cases compared to results from studies using other valuation techniques to arrive at a complete measure of value.

Travel Cost

The basis of the travel cost approach is that the use of ESS may require travel. The travel costs incurred to enjoy those ESS can be seen as a reflection of the willingness to pay for those services, reflecting the implied value of the services (de Groot et al., 2002; Farber et al., 2002; Farber et al., 2006; Turner et al., 2010). However, difficulties occur when considering the point of origin for visitors. Some visitors may be local while others live farther away, thus incurring different travel cost values. Additionally, multipurpose trips and defining and measuring the opportunity cost of time add further complications to the application of this method.

There has been very limited application of this approach to water ecosystems. The approach is limited to direct use recreational benefits and typically has been applied in the cases of recreational areas, national parks and ecotourism facilities. In many cases, the applicability to urban water cycle valuation seems therefore limited.

Some argue that the travel cost approach can be used to determine water-related recreational values of water reservoirs such as boating, angling and general visiting and to determine water-related recreational values of wetlands such bird watching and general visiting (Boyer and Polasky 2004). However, the travel cost approach only evaluates part of the total value of water ecosystems and cannot be used to value their respective public goods aspects (e.g. flood control, ground water recharge and discharge) that are unrelated to recreation.

Hedonic pricing

Hedonic pricing method (HPM) relies on the theorem that the value an individual places on a service is based on the attributes it possesses (Chee, 2004) and that the service demand may be reflected in the willingness to pay/accept for associated goods (de Groot et al., 2002; Farber et al., 2002; Farber et al., 2006; Turner et al., 2010). In that regard, the economic value of a characteristic of the service can be derived from the market price of the service (Chee, 2004).

The main application of this method is to estimate the willingness to pay for real estate. According to Palmquist (2005) property value studies are one of the most frequently applied techniques for benefit measurement, as one of the only places where environmental quality is traded on explicit markets is for real estate. However, problems arise from the fact that hedonic pricing relies on the underlying assumption that property prices are sensitive to the quality and provision of ESS. Realistically, property markets are not perfectly competitive and ecosystem quality and supply are not the only characteristics of where people buy real estate. It is difficult to isolate specific ecosystem effects from other determinants of property prices and accurate statistical inference must be done in order just to identify the relation between homes prices and ESS presence.

There are only a few hedonic pricing studies dealing with water quality in the environmental economics literature (Leggett and Bockstael, 2000; Springate-Baginski et al., 2009; Steinnes, 1992). This is because many water quality indices measure pollutants that are impossible for residents to observe or that do not directly impair the enjoyment the individual derives from his/her waterfront home. People only recognise perceptible changes, limiting the method to capturing people's willingness to pay for perceived differences in environmental attributes, and their direct consequences (Leggett and Bockstael, 2000). Thus, if people aren't aware of the linkages between the environmental attribute and benefits to them or their property, the value will not be reflected in home prices (Springate-Baginski et al., 2009).

The approach may only capture direct-use values of water-related ESS as perceived by the consumers of the good, who are the (nearby) property owners. Services such as flood control, water-quality improvement, habitat provision for species, and ground-water recharge and discharge, may provide values that accrue far away to individuals other than local property owners. If so, HPM will not accurately capture the full value of services provided.

Lastly, the application of HPM to water-related ESS, and a weakness in this technique, is the very large data sets and detailed information that must be collected, covering all of the principal features affecting prices (Springate-Baginski et al., 2009).

Stated preference methods

In stated preference (SP) survey respondents are asked questions that embody information about social preferences. Here, hypothetical markets are introduced and respondents have to define a value, in different ways, for the respective ESS within these markets (Spangenberg and Settele, 2010).

SP approaches are criticised for overlooking concerns about procedural justice, non-utilitarian ethics and the role of social norms (Lo and Spash, 2013). Therefore “social value” approaches were introduced, a classification which contrast the role of individuals versus groups in the process of valuation and differentiates between individual and social values as products of any such process (Spash, 2007).

Contingent valuation

According to Carson and Louviere (2011) contingent valuation conveys three main elements: (1) information related to preferences is obtained using an SP survey, (2) the study’s purpose is placing an economic value on a good, and (3) the good being valued is a public one. One elicitation methods is a matching approach, where respondents are asked to provide a number (their willingness to pay or willingness to accept compensation) that will make them indifferent in some sense.

Another elicitation method is discrete choice experiments, where respondents pick their most preferred alternative from a set of options. Respondents are asked to make a discrete choice between two or more alternatives in a choice set, where the alternatives presented are constructed by means of an experimental design that varies one or more attributes within- and/or between-respondents to be able to estimate economic quantities tied to preference parameters (Carson and Louviere 2011).

Independent of the chosen method, it is important to recognise that contingent valuation can lead to certain types of bias within the survey results: operational, hypothetical, information, design and strategic bias (amongst others) (Mitchell and Carson, 2013). Therefore, careful consideration must be placed into the design and conduction of the survey in question.

Despite the challenges posed in addressing the numerous types of bias, almost any ESS can potentially be valued with the application of contingent valuation approaches (de Groot et al. 2002). Examples include the willingness to pay for increases in water quality, fishing improvement conditions, flood protection, wetland habitat and services preservation (Boyer and Polasky, 2004). More importantly, stated preference methods like contingent valuation are the only approaches available for the valuation of non-use values of water-related ESS. These include existence values like the enjoyment of seascapes; and bequest values like the willingness to preserve water ecosystems for the experience and use of future generations.

Benefits transfer environmental valuation method

Due to time and financial constraints, some studies employ the valuation results of other primary studies to predict welfare estimates for other sites of policy significance for which primary valuation estimates are difficult to attain or are unavailable (Johnston and Rosenberger, 2010). The benefits transfer method ranges in form from unit-value or point-estimate transfers, function transfers and meta-analytical approaches that synthesise results of numerous studies deemed somewhat related to the study in question (Iovanna and Griffiths, 2006).

In general, a consensus of the literature suggests that function transfers typically outperform unit-value transfers as they attempt to calibrate value estimates to the study site in question through population and socio-demographic adjustments (Rosenberger and Stanley, 2006). However, critics of this approach caution the inherit flaws of this method due to the fact that the characteristics of the consumers or client group for which data exist may differ from those of the transfer site. These factors can limit the extent to which values can be transferred or generalised (HM Treasury, 2003). Additionally, the meta-analysis approach is based on studies that a researcher deemed 'somewhat related' to the transfer site, calling in question subjective bias of the studies included.

Despite criticism of this approach, there is an increase in the use of benefits transfer method as primary valuation databases expand and more sophisticated benefits transfer methods are generated (HM Treasury 2003; Johnston and Rosenberger 2010). Though primary research is generally preferred to estimate ESS values, policy processes and financial limitations often dictate that benefit transfer is the only feasible solution (Johnston and Rosenberger 2010; Iovanna and Griffiths 2006). For example, the EU WFD mandates the consideration of benefits and costs for river basin management, including many large and small water bodies across multiple countries. This mandate has encouraged the increasing use of the benefits transfer method as a cost-effective means of benefit estimation (Hanley et al., 2006a; 2006b).

Benefits transfer has been applied in numerous water-related ESS valuation cases at varying levels of scale (Iovanna and Griffiths, 2006; Desvousges et al., 1992; Johnston and Rosenberger, 2010). Examples of water-related benefits transfer range from increases in fish populations, recreation benefit of contaminant-free fishing, changes in water provisioning service flows, water quality improvements, and willingness to pay for flood control and wetland conservation (Iovanna and Griffiths, 2006; Brouwer and Bateman, 2005).

Valuation barriers and limitations

The inherent uncertainties and lack of agreement surrounding the practical application of ESA have resulted in different understandings and adaptations of the approach. While it is considered controversial by some actors, expressing the value of nature and its services in monetary terms is key to the approach and intends to promote better informed decision-making. As the vast majority of ESS have no market price, price and value estimations must be obtained using alternative methods. The multiplicity of options to do this and the lack of a standard procedure for choosing between them raise the already high level of complexity and abstraction of the discussion surrounding ESA.

A number of general barriers exist which are commonly encountered by scientists and practitioners conducting economic valuation of ESS. Due to their visibility and policy relevance, large-scale collaboration projects like DESSIN represent great opportunities to address these issues. Some of these general barriers are listed and explained here with the intention of providing the context from which this report will go forward and propose a plan of action.

- **Inconsistent definitions/conflicting typologies:** there are diverse or even conflicting meanings for various environmental valuation methods found in the existing literature (i.e. Carson and Louviere, 2011). Different authors may employ different underlying assumptions and typologies to classify the methods used to assess the value of ESS. This development has rendered the comparison of the individual strengths and weaknesses of the different methods a highly complex task, at a time when the current economic/environmental setting demands higher efficiency and reliability in the practical application of the ESA.
- **Unjustified preference/attention to a certain method/ESS type:** ESS valuation methods are extensively used in the production of academic literature. However, some methods have few existing applications (especially for water related ESS). This may result in a great deal of literature available on a specific type of service or methodology, while knowledge and progress on others is limited.
- **Combination of different methodologies in a single assessment:** obtaining a quantitative figure to measure ecosystem change following the total economic value approach can easily lead to issues of aggregation of different concepts of value (coming from the application of different methodologies applied to different services), double counting, substitution, etc. In some cases, two valuation methods are mixed, combined, or used in parallel without any clear distinctions being made.
- **Lack of benchmark studies/practically applied methods:** there is a lack of complete educational case studies that can be used as benchmarks to explain the whole sequence of the valuation process: data sources and data mining, selection of method(s), application, validation of results and discussion. The main challenge lies in the applicability of environmental valuation methods in real management, as most project assessments are only based on cost-benefit analyses and do not consider environmental externalities.
- **Lack of data:** a situation where accessibility to accurate, high resolution data that is relevant to the site/ecosystem being assessed is low can greatly limit the process of economic valuation, with only small distinctions depending on the method used.

In light of the multiple obstacles to the successful implementation of environmental valuation methodologies, some potential solutions are proposed. Firstly, conduct a review of current environmental valuation methodologies to set the stage and ensure that no relevant knowledge has been overlooked. Secondly, develop harmonised definitions of concepts and terminology that fosters common understanding and collaboration between different sectors (e.g. economists and ecologists). Concurrently, develop a demonstrated/validated environmental valuation approach with a unified classification of methodologies at its core. More precise definitions and a common classification of the existing methods could serve to overcome the complexities commonly related with conducting integrated assessments of the services provided by ecosystems.

Optimally, the solutions proposed above should be complemented by clear guidance on how to select an appropriate valuation method for a specific type of ESS. This would necessitate the identification of relevant and suitable indicators to measure changes in ESS provision in relation to specific changes in ecosystem status. In turn, such indicators should be relevant for current policy targets and priorities. In this regard, sharing experiences with other related EU projects and initiatives would facilitate the calibration and validation of outcomes and ensure their practical application at a wider scale. The use of real case studies should help to increase the understanding of the various methods, their underlying assumptions and their possibilities and limitations.

Classification/typology of valuation methods

As mentioned above, the lack of a common approach for the classification of valuation methods has resulted in the surfacing of many different typologies. DESSIN proposes a very simplified typology shown in Figure 11 above, using the most consistent elements of the discussed typologies.

6. How to incorporate sustainability assessment into the DSS of DESSIN

The last task within WP11.1 is about linking the ecosystem assessment to a sustainability assessment in order to provide a full-scale perspective on ecosystem service valuation from both sustainability assessment and ecosystem valuation perspectives. As the topics of ESS and ESS provision were already extensively described in the previous chapters, the following deliberations will mainly focus on sustainability assessments and its linkage to ecosystem service assessments. Firstly, an overview of the concept of sustainability is provided. There are two concepts related to it: *sustainable development,* that refers to actions taken to achieve a sustainable level in a given time horizon and *sustainability,* that refers to a current status. Preliminary ideas on how a sustainability assessment might be combined with the assessment of sustainable usage of ESS are given. A short overview of the sustainability assessment approach developed in the EU-TRUST project for urban water cycle systems is presented. Tools and methodologies developed in TRUST to assess sustainability are also briefly described, as examples of tools for evaluating changes in sustainability via new technologies.

6.1 Sustainability

The World Commission on Environment and Development (WCED) conference in 1987, which took place at the end of an initiative for elaborating a report concerning the perspectives of a worldwide sustainable and environmentally sound development up to the year 2000 and beyond, provided the first and so far most-referred definition of the term *Sustainable Development*. In the final report on the said conference, “Our Common Future”, also known as “Brundtland Report”, it says: “Sustainable development seeks to meet the needs and aspirations of the present without compromising the ability to meet those of the future” (WCED, 1987). Sustainable development therefore is “a process of change in which the exploitation of resources, the direction of investments, the orientation of technological development; and institutional change are all in harmony and enhance both current and future potential to meet human needs and aspirations” (WCED, 1987). Since then, several attempts were made to define sustainability (Olschewski and Klein, 2011; Wu, 2013). According to Elkington (1998), for example, sustainability can be seen as the social, environmental and economic qualities of a given system under study, in a holistic and long-term perspective. These three dimensions of sustainability, also known as triple bottom line (TBL) (see Figure 12), are interdependent so that they always are balanced (Figge and Hahn, 2004).

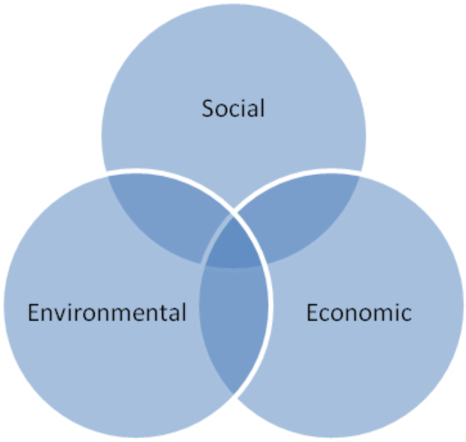

Figure 12. Triple bottom line of sustainability

Based on this definition of sustainability, the available literature provides studies on how the interdependencies between these dimensions can be described. Some accept compensation between the dimensions, others do not. The concept of *weak sustainability*, for example, suggests that “a system is considered sustainable as long as its total capital increases or remains the same”(Wu, 2013). To put it simply, this means that a region where the environmental quality is neglected in favour of a rapid economic development and urban sprawl is considered as sustainable as long as the same level of welfare is assured (Hartwick, 2000; Wu, 2013). This concept has the inherent danger of leading to overexploitation of natural resources and extensively using ESS as long as they seem to lead to economic growth. In contrast, *absurdly strong sustainability* advises to conserve nature at its present state and implies not to use any non-renewable resource at all. This implies that humans shall not use finite resources like oil, gas, etc. in favour of economic growth and human welfare. Also, this means there is no substitution between natural capital and man-made capital (e.g. by using non-renewable resources to produce any kind of goods) allowed (Khalili, 2011; Holland, 2002; Daly et al., 1995). Thus, in terms of the TBL explained above, compensation towards economic state while degrading the environmental state in terms of depleting resources is a “no go”. Regarding the use of ESS this would imply, for instance, that a water utility may use an aquifer as a drinking water resource as long as the water taken does not exceed the natural ground water recharge in a given timeframe. However, the water utility would not be allowed to pump this water from the ground using electric energy, since this energy could hardly be produced without using at least a specific proportion on non-renewable resources (coal, etc.). The concept of *strong sustainability* proposed by Daly (1995) is in between these extreme positions. “Strong sustainability assumes that man-made and natural capital are basically complements”(Daly et al., 1995), whereby the economic component is part of the social domain with the environment as the constraining factor all around (Wu, 2013).

6.2 Sustainability and ESS

As a reaction on the supranational developments described above, sustainable development and responsible dealing with ecosystems have already been included in the EU WFD (2000/60/EC).

Within this directive, the responsible management of ecosystems is highlighted in order to ensure sustainability in the social use of water and water environments (Sørensen et al., 2014). For that and other reasons, changes in the way utilities use ecosystems are unavoidable (Wu, 2013).

However, the main focus of the DESSIN project is on improving ecosystem state by applying selected technologies. In the context of sustainability, deciding on the benefits of these technologies may be very difficult due to conflicting interests. For example, from one point of view (e.g. technical performance), a certain technological innovation can be seen as a major improvement in comparison to the state-of-the-art technology, while implying increasing operating costs or social impact to ESS users on the other hand. Therefore, the valuation of benefits for different technical systems or alternative solutions in general can be very complex. The reason for this is that implementing a new technology that causes changes in ESS provision might imply trade-offs between different sustainability dimensions. In order to demonstrate awareness of these challenges, establishing a comprehensive framework to illustrate the correlations between changes in ESS provision and their effects on the dimensions of sustainability is needed (Sørensen et al., 2014). Thus in order to assess the impact of new technologies, the TRUST sustainability assessment approach will be taken into account, since it broadens the scope.

6.3 Sustainable use of ESS

The main focus of D11.1 is to give an internal state-of-the-art report on ecosystem service evaluation regarding water status related changes in ESS, economic valuation of ESS and sustainability assessment of impacts brought by the demonstrated technologies. The proposed DPSIR framework to integrate these single aspects has already been described in Chapter 1.

Within this framework, an ecosystem has a state which implies a certain capacity to deliver ESS to humans that may cause subsequent benefits to human well-being. The services that the ecosystem provides and which society uses bring a certain benefit to society. This benefit may be valued (sometimes also in monetary terms). Society might feel the need to have an influence on ESS provision, triggered by a specific value perception. For example, perceived flood risk might lead to decisions to implement flood risk reduction measures. Driving forces influencing river basin management might lead to specific actions. For instance, by restoring wetlands, the state of the ecosystem can be changed. By doing so, the ecosystem has a greater capacity to deliver the regulating service of flood protection, which in turn leads to a higher benefit. A water utility aims to deliver its services to customers in a sustainable way. Innovations, like new technologies implemented, will alter ecosystems. At the same time, those innovations should prove to be sustainable in terms of the utility's way of serving its customers.

DESSIN is focusing on the ecosystem itself and its biophysical basis. The TRUST project focuses more on water utilities (see below in section 6.4). The utilities are fulfilling humans' needs by anticipating their value perceptions for services like drinking water provision, waste water treatment, etc. The actions necessary for this do, of course, influence the ecological state of the water bodies involved in the "production chain" of water utilities. The question now is how to link sustainable use of ecosystems by humans (DESSIN) with sustainable urban water services delivery as highlighted in TRUST.

Therefore, there is a need to define how to interpret sustainability from DESSIN's point of view. Hence, it is necessary to differentiate between ESS capacity, actual provision of ESS, human need for these services and the actual use of these services ("flow"; Schröter et al., 2014). According to Schröter et al. (2014) it is especially necessary to distinguish between the capacity to provide services and their actual use. Moreover, the ecosystem's capacity equals the potential of a specific ecosystem to provide certain services and goods (Burkhard et al., 2012); therefore limiting the extent to which a sustainable use of services is possible. This limit is aligned to the current management of the ecosystem. Thus, deviations in ESS provision caused by modified human behaviour or innovations might change this. Consequently, the actual use of ESS determines whether the ecosystem is treated sustainably or not. This use is interlinked to preferences like costs of use, availability and biophysical characteristics (Schröter et al., 2014). Following this logic, we can define sustainable use of ESS in DESSIN as follows:

- Sustainable use of ecosystem:
  ESS use <= Ecosystem capacity
- Non-sustainable use of ecosystem:
  ESS use > Ecosystem capacity

This is valid for any given ecosystem and its capacity[26] to deliver any ecosystem service for a given time frame, e.g., one year. However, as this is a simplified definition, there might be more complex situations where this equation is not sufficient to define whether the use of an ecosystem is sustainable or not over time. For instance, in the short term, it may be true that use of an ecosystem service does not exceed capacity, while in the longer term use might do so.

Now having defined sustainability for DESSIN's purpose in a generic way, the next section summarises the TRUST approach. Following that, a means to use the TRUST approach to assess if the implementation of a new technology (as to be developed in DESSIN) is sustainable or not, is given.

6.4 The TRUST project and its contribution to the assessment of "sustainable services"

TRansitions to the Urban Water Services of Tomorrow (TRUST)[27] is a four year (2011-2015) research project funded by the European Union (7FP). The ambition of TRUST is to deliver co-produced knowledge to enable water utilities to achieve a sustainable future urban water system without compromising service quality. The focus of TRUST is on delivering products, methodologies and solutions to ensure "sustainable water service" and sustainable development of the water service

[26] Side note: In contrast to chapter 3 it is referred to „capacity" vs "use" here. This is because in terms of a sustainability assessment it is needed to check whether the use of ecosystem goods/services exceeds its inherent ability to serve these goods/services on a given time frame without compromising future potential to do so.
[27] For more information visit: www.trust-i.net/

while taking into account present and future challenges (i.e. climate changes, population growth, ageing, etc.).

As described above, the concept of sustainability is frequently associated with the triple bottom line (TBL) approach, comprised by social, environmental and economic dimensions. These dimensions can be seen as aggregated objectives relative to a particular sector that should be developed. This begs the question of whether the TBL approach is the most appropriate to assess the sustainability of a public service (e.g. water service).

TRUST concludes that “the TBL approach is not enough to characterise the urban water cycle services sustainability since technical (infrastructural) and governance aspects are also quite relevant. Even if they are not one end in themselves, they are instrumental and essential for the social, environmental and economic dimensions and objectives of sustainability” (Marques and van Leeuwen, 2012).

The infrastructural dimension, as the name suggests, is associated with the system of physical infrastructure and might encompass aspects concerning the system performance, its durability, reliability, flexibility and adaptability, and, among other aspects, are quite associated with asset management.

Governance is related to the rules of the game, the respect for those rules by the stakeholders, the transparency, their participation in the decision making process, particularly the customers, the effectiveness and efficiency of the measures taken and the quality of the accountability and adjustment mechanisms. The existence and alignment of city planning with the urban water cycle services is also a relevant governance issue.

Thus, according to TRUST, specific targets must be met in the two supporting dimensions as well to ensure sustainability (in addition to the TBL).

The aim of this section is to present the approach developed in TRUST to define, assess and measure sustainability of an urban water service in order to build and adopt within DESSIN a similar methodology to analyse the sustainability of the ecosystem service.

Sustainable urban water service - definition as for TRUST

TRUST research builds on the definition of "sustainability of the urban water cycle service" (UWCS), which is met when the quality of assets and governance of the services is sufficient to actively secure the water sector’s needed contributions to urban social, environmental and economic development in a way that meets the needs of the present without compromising the ability of future generations to meet their own needs" (Alegre et al., 2014).

The TRUST definition of sustainability of water service, in addition to the traditional triple bottom line dimensions, has put emphasis on asset reliability and governance involvement. Social, environmental and economic sustainability are still the *main* dimensions, with a further two dimensions (assets and governance) as required *supporting* dimensions.

From the TRUST perspective, focusing on the water service delivered, main dimensions (social, environmental and economic) are seen as the *end dimensions* of decisions and interventions towards sustainability, for which the urban water cycle services of any city should strive for a balanced high quality. The supporting dimensions are the ones that a water utility or a city can directly influence, by transition policies and interventions with respect to either governance or assets management. These policies and interventions should, of course, also aim to improve the ends of social, environmental and economic sustainability.

Sustainability assessment based on TRUST definition

A set of sustainability objectives and criteria are identified to assess the level of sustainability of a given service. The assessment is made in practice by examining a chosen set of performance metrics/indicators and how they comply with the predefined objectives and criteria.

Setting up objectives, assessment criteria, metrics and targets is a crucial stage in order to set up clear directions of action, as well as accountability of results through timely review.

This sequence shall be followed to establish the TRUST sustainability assessment. Clarifying the four distinct but sequential concepts:

Objectives are the goals that the organisation aims to achieve. According with the ISO 24510:2007 (ISO, 2007a), 24511:2007 (ISO, 2007b), 24512:2007 (ISO, 2007c) standards, the TRUST performance assessment should always be linked to objectives that are clear and concise, as well as ambitious, feasible and compatible, and take into account the ultimate goal for the utility to provide a sustainable service to society. For each objective, it is recommended that key assessment criteria be specified.

Assessment criteria are points of view that allow for the assessment of the objectives. For each criterion, performance, risk and cost metrics must be selected to set clear targets, and for further monitoring of the results.

Metrics are the specific parameters or functions used to quantitatively or qualitatively assess criteria; metrics can be indicators, indices or levels.

Targets are the actual proposed values to be achieved for each metric within a given time frame (short, medium or long term).

For each objective, one should develop a description and targets for the desired situation in a given year (2040 is the reference time horizon in TRUST), and the given assessment criteria provide a basis for developing and selecting indicators and indices for the measure of quality in how the urban water cycle services and their various components perform. The performance metrics/indicators may be quantitative and/or qualitative, and are chosen to take account of the particular context and challenges of a given urban water cycle system, in a medium- and long-term transition context.

The list of objectives and criteria selected in TRUST to assess the sustainability level of an Urban Water System (UWS) is provided below (Table 6), however the list may not be suitable for all purposes, therefore the metrics can also be user defined.

Table 6. Dimension, objectives and criteria of the urban water cycle system sustainability

Sustainability Dimension	Sustainability Objective	Sustainability Criteria
Social (S)	S1) Access to urban water services	S11) Service coverage
	S2) Effectively satisfy the current users' needs and expectations	S21) Quality of service
		S22) Safety and health
	S3) Acceptance and awareness of UWCS	S31) Affordability
Environment (En)	En1) Efficient use of water, energy and materials	En11) Efficiency in the use of water (including final uses)
		En12) Efficiency in the use of energy
		En13) Efficiency in the use of materials
	En2) Minimisation of other environmental impacts	En21) Environmental efficiency (resource exploitation and life cycle emissions to water, air and soil)
Economic (Ec)	Ec1) Ensure economic sustainability of the UWCS	Ec11) Cost recovery and reinvestment in UWCS (incl. cost financing)
		Ec12) Economic efficiency
		Ec13) Leverage (degree of indebtedness)
		Ec14) Willingness to pay (accounts receivable)
Governance (G)	G1) Public participation	G11) Participation initiatives
	G2) Transparency and accountability	G21) Availability of information and public disclosure
		G22) Availability of mechanisms of accountability

	G3) Clearness, steadiness and measurability of the UWCS policies	G31) Clearness, steadiness and measurability of policies
	G4) Alignment of city, corporate and water resources planning	G41) Degree of alignment of city, corporate and water resources planning
Assets (A)	A1) Infrastructure reliability, adequacy and resilience	A11) Adequacy of the rehabilitation rate
		A12) Reliability and failures
		A13) Adequacy of infrastructural capacity
		A14) Adaptability to changes (e.g. climate change adaptation)
	A2) Human capital	A21) Adequacy of training, capacity building and knowledge transfer
	A3) Information and knowledge management	A31) Quality of the information and of the knowledge management system

TRUST products for sustainability assessment

TRUST offers a range of ready-to-use tools for water utilities and other stakeholders of urban water cycles to assess whether their behaviour (e.g. service delivering to consumers in a specific way) is sustainable or not.

The TRUST tools can be classified in those aiming at estimating the single metrics or indicators that are then inputs to those tools aiming at estimating the overall sustainability level according to set targets for selected objectives. Some relevant examples are given below.[28]

Tools to compute individual indicators

WaterMet2 (Behzadian et al., 2014) - The model WaterMet2 has been developed to support the framing and measuring of indicators for assessing the degree of improvement in terms of the performance of the urban water system in the five sustainability dimensions. WaterMet2 is a conceptual, simulation type, mass-balance-based, integrated UWS model which quantifies metabolism-related key performance of UWS with focus on sustainability-related issues. Metabolism in UWS refers to the fluxes and conversion processes related to all kinds of water flows, materials and energy in the UWS, which are necessary to fulfill the necessary functions (Venkatesh and Brattebø, 2011).

[28] More specific tools can be found at http://www.trust-i.net/project/index.php.

Dynamic Metabolism Model (DMM) (Venkatesh et al., 2011) – as well as WaterMet2, DMM is built upon the concept of quantifying the metabolism of UWCS, with the TRUST metabolism modelling system definition and scoping report as a starting point (Venkatesh and Brattebø, 2011). It is used to quantify selected indicators to analyse flows and conversion processes of all kinds of materials and energy, which are mobilised by the development and operations of the system in order to fulfill the necessary functions – water supply and sanitation at given quantity and quality levels. The adopted methodology consists of the computation of selected sustainability indicators with DMM and then in comparing their variation during the period of simulation from the current value for different scenarios of interventions. DMM is a simple, user-friendly Excel-based model that, started off as a model inspired by industrial ecology and is anchored on the environmental platform (with a strong focus on the material stocks and flows, energy consumption and the associated environmental impacts).

Risk management approach (Ugarelli et al., 2014) - In TRUST, risk assessment is applied to risks related to potential impacts in systems due to changes in circumstances using a strategic level and, therefore, an aggregative approach. The methodology proposed essentially follows the standard steps of a risk management process as defined by ISO 31000:2009 (ISO, 2009) , but adjusted to be used at strategic (macro) level using an integrated approach. Assuming established sustainability objectives defined for a specific system, risks can be identified in the context of occurrence of circumstances as events causing undesired and uncertain deviations from the objectives (risk defined as effect of uncertainty on objectives in ISO (2009)). In each specific application, the objectives need to be expressed by an appropriate set of criteria, supported by appropriate metrics and corresponding targets. The deviations from the expected situation in relation to the set targets, resulting from the occurrence of the undesired circumstances, are the corresponding consequences.

Tools to assess the sustainability level

Financial Sustainability Rating Tool (FSRT) (Hoffjan et al., 2014) - The FSRT offers water supply and/or waste water removal companies an opportunity to rate the utility's financial sustainability. It gives the user an indication of which area, from financial situation over asset management to business operation, needs optimisation. The Tool also evaluates different forecasts (e.g. population development) and country specific characteristics (e.g. inflation rate) to assess future trends. It needs quantitative data input and is an assessment if a utility produces its services "cost-covering".[29]

The TRUST self-assessment tool (Alegre et al., 2014) – it is an easy-to-use assessment tool that, following the TRUST sustainability definition, provides utilities with a first glimpse of readiness towards the 2040 target.[30]

[29] To access the tool and for more information visit: http://fsrt-trust.ing. unibo.it/fsrt
[30] To access the tool and for more information visit: http://self-assessment.trust-i.net

Decision support system for the long-term city metabolism planning problem (Morley et al., 2014) – it is a DSS tool for the assessment of intervention strategies in an UWS. Lists of intervention options and Performance Indicators are exposed by the DSS for the user to identify one or more optimal intervention strategies from a list of pre-defined strategies over a fixed long-term planning horizon using evaluation criteria mapped to the TRUST sustainability criteria. A Multi-Criteria Decision Analysis approach is employed in the DSS to compare the defined intervention strategies and rank them with respect to a pre-specified weighting scheme for different scenarios. A rich, interactive Graphical User Interface is employed to assist with guiding the end user through the stages of defining the problem, evaluating and ranking intervention strategies as well as providing novel visualisation tools for comparing solutions. This mechanism provides a useful tool for decision makers to compare different strategies for the planning of UWS with respect to multiple scenarios.

How to adapt the TRUST sustainability assessment methodology to DESSIN?

Elements of the TRUST assessment approach might be useful to assess whether the implementation of (innovative) technologies alters sustainability. This includes checking the situation before and after implementation from the water utility's point of view. This is especially to be investigated for the following groups:

- State and capacity of the ecosystem
- Actual provision of a specific ecosystem service
- Actual use of a specific ecosystem service
- Human need for a specific ecosystem service (and its use in terms of water utilities delivering their services to consumers)
- Valuation of these ESS in monetary or non-monetary units
- Impacts on other social aspects
- Impacts on other costs
- Impacts on governance (image of the water utility, etc.)

To assess the sustainability of an implemented DESSIN technology, the use of the five TRUST dimensions may be helpful. The environment dimension might be assessed by using the generic assessment idea as presented in section 6.3. In addition, fitting environmental sustainability indicators from TRUST could be used. The other four dimensions can also be assessed by adapting indicators from TRUST. This is especially valuable, since it widens the scope between human well-being, societal response and driving forces. Critical trade-offs of DESSIN's new case studies to foster innovative technologies will be more easily identified using TRUST's comprehensive sustainability framework. This is because it can build on the biophysical basics becoming obvious once checking sustainability from DESSIN´s point of view in the environmental dimension and linking that with a given set of indicators for the social, economic, governance and assets dimension.

7. Next Steps

As mentioned earlier, the main objective of this document has been to bring the DESSIN consortium up to speed on the latest developments of the ESS discussion, setting the foundations for future work within WA1. So far, this has resulted in the consolidation of ESS terminology to be used within the project, the selection of a conceptual approach upon which the ESS Evaluation Framework will be built (and its adaptation to fit the needs of DESSIN), the selection of a common classification of ESS to be used as a practical basis for our framework, and initial discussions on critical issues for this Work Area that have not yet been resolved by current state-of-the-art work (e.g. linking changes in ecosystem state to changes in ESS, estimating changes in ESS based on technological implementations, accounting for temporal and spatial aspects of ESS, etc.). From this standpoint, the following actions are foreseen for the second year of work under WA1:

- Identification and selection of suitable indicators/proxies to describe relevant ESS (freshwater ESS related to quality and quantity)
    - WA1 partners will agree on a set of criteria for the identification and selection of indicators/proxies. Some of the criteria proposed so far include policy relevance, ecological and methodological soundness, data availability, coverage of the complexity of the studied ecosystems and linkages with the selected ecosystem elements, among others.
- Selection of an analytical framework to evaluate changes in ESS that is most appropriate for the needs of WA1
    - An analytical framework for ESS assessment will be chosen to underpin the comparison between a technology's implementation costs and its benefits in terms of impact on ESS. The choice will take into consideration the appropriateness of the related indicators for ESS assessment, issues of spatial and temporal scale, trade-offs, double-counting and uncertainties, among others.
- Preparation of a first version of the evaluation methodology for testing at mature sites
    - The preliminary version will be developed on the basis of the CICES classification and will provide a schedule and guidance on qualitative and quantitative assessments followed by monetary valuation of selected freshwater ESS.
- Improvement of the conceptual approach and the first evaluation methodology into a final version
    - A final, fine-tuned and validated version of the conceptual approach and methodology will be developed based on the practical recommendations streaming from the tests on the mature sites. This will consider data requirements and specifications that will support the development of the ESS module in WP23.

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The research leading to these results has received funding from the European Union Seventh Framework Programme (FP7/2007-2013) under grant agreement no. 619039
This publication reflects only the author's views and the European Union is not liable for any use that may be made of the information contained therein.

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