A portfolio of ideal types of (public and private) governance modes for selected ES/NC

Torsten Krause[a], Pontus Ambros[a] and Lennart Olsson[a]
with input from Deirdre Joyce[b]; Ana Ruiz[c]; Anja Liski[d], Marianne Kettunen[e]
[a] Lund University; [b] University College Dublin; [c] Institut Mediterrani d'Estudis Avançats; [d] University of Edinburgh; [e] Institute for European Environmental Policy

November 2016

OPERAs

Ecosystem Science for Policy & Practice

This project has received funding from the European Union's Seventh Framework Programme for research, technological development and demonstration under grant agreement number 308393.

Prepared under contract from the European Commission
Project reference: 308393
Collaborative project
FP7 Environment

Project acronym: **OPERAs**
Project full title: **Operational Potential of Ecosystem Research Applications**
Start of the project: 1 December 2012
Duration: 54 months
Project coordinator: The University of Edinburgh
Project website: operas-project.eu

Title: A portfolio of ideal types of (public and private) governance modes for selected ES/NC
Deliverable number: 3.6
Nature of the deliverable: Report
Work package responsible: WP3
Partner responsible: ULUND
Other partners involved: University College Dublin; Institut Mediterrani d'Estudis Avançats; University of Edinburgh; Institute for European Environmental Policy
Deliverable status: Final

Version	Status	Date	Authors
1.0	Final	November 2016	Torsten Krause, Pontus Ambros, Lennart Olsson - Lund University Centre for Sustainability Studies

This project has received funding from the European Union's Seventh Framework Programme for research, technological development and demonstration under grant agreement number 308393.

Contributing authors and institutions

Lund University Centre for Sustainability Studies, Lund University, Sweden
Torsten Krause, Lennart Olsson and Pontus Ambros

University of Edinburgh, Edinburgh, Scotland
Anja Helena Liski

Institut Mediterrani d'Estudis Avançats (IMEDEA), Esporles, Mallorca, Spain
Ana Ruiz

School of Architecture, Planning and Environmental Policy, University College Dublin (UCD), Ireland
Deirdre Joyce

Institute for European Environmental Policy (IEEP), Brussels, Belgium
Marianne Kettunen

Acknowledgements

The research leading to the results in this deliverable has received funding from the European Commission Seventh Framework Programme under Grant Agreement No. FP7-ENV-2012-308393-2 (OPERAs).

Contents

List of Tables

List of Figures

1. Introduction

In response to observed and projected global environmental changes governance for the sustainable use and management of ecosystems and ecosystem services at the national, regional and global levels is becoming ever more important in order to strive for sustainability and more sustainable development. Within environmental governance research and practice, various approaches to tackle global environmental problems are discussed, tested and implemented, ranging from global agreements, such as UN conventions, to regional, national and local mechanisms for conservation and exploitation of nature. Around the world, there are both successful (e.g. Montreal Protocol) and unsuccessful (e.g. the Convention on International Trade in Endangered Species of Wild Fauna and Flora, CITES (Bennett, 2015) examples of how environmental issues have been tackled through environmental governance.

The ecosystem services concept first appeared in the early 1980's (Ehrlich and Mooney, 1983), with iterations and various definitions arising since then. Most of these have in common that the notion of services clearly delineates the anthropocentrism and the utilitarian framing of those ecosystem functions which are useful for humans (Braat and de Groot, 2012). Based on this re-alignment of environmental governance towards an inclusion of externalities and the linkage with economic cost-benefit analysis, the use of the ecosystem services concept gained ground within and outside of academia (TEEB, 2010, Hauck et al., 2013, TRUCOST and TEEB, 2013). Nevertheless, even though there are many attempts to categorize ecosystem services and group these according to their properties (De Groot et al., 2002) or how they affect human well-being (MEA, 2005), ecosystem services by and large are a complex and interlinked set of ecological processes (de Groot et al., 2010, Muradian and Rival, 2012). These complex and interlinked set of ecological processes contribute to livelihoods at different spatial scales and through varying combinations (Willemen et al., 2013).

The ES/NC concept is supposed to strengthen thinking in systems—not only in terms of ecosystem processes and functions, but also with regard to social and political systems—and emphasizes the linkages between ecological and human systems (Costanza et al., 1997). Yet, the concept of ecosystem services should be seen as a boundary object through which dialogue and cooperation between economists and ecologists, and between scientists and policy makers can be inspired. Because of its interpretive flexibility the concept is claimed to facilitate transdisciplinary research processes (Schröter et al., 2014). However, even though there is a vivid and critical scholarly debate about the ecosystem services and natural capital concepts and its usefulness as well as pitfalls (Spash, 2009, Gómez-Baggethun and Ruiz-Pérez, 2011, Luck et al., 2012, Jax et al., 2013, Norgaard, 2010), both are claimed to be valuable concepts when deciding how to allocate the resources provided by nature among alternative desirable ends (Farley, 2012, Schröter et al., 2014).

This, however, comes with another range of issues. Native ecosystems can be replaced by for example a plantation forest, which more effectively delivers carbon storage and maintains the hydrological services. Thus, there is an increasing recognition that we must not rely on a narrow ecosystem-services approach that includes only single or few ES, because it misses out on many other values as well as ecosystem functions, biodiversity and supporting services, which in the narrow sense might not

immediately be useful or valued by society (Odling-Smee, 2005). ES can be described as a boundary object that links underlying ecosystem functions to social goods and services (Figure 1).

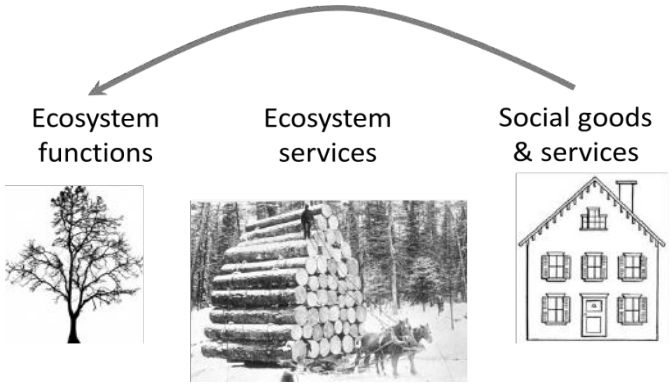

Figure 1 – Representation of ecosystem services as a boundary object that links ecosystem functions to associated social goods and services

Nowadays, the Ecosystem Services framework is increasingly being used in research (Seppelt et al., 2011) and has been taken up by policy-makers, the business sector and international non-governmental organizations (TRUCOST and TEEB, 2013). In the past years, the classification of services has changed substantially. In the OPERAs project the Common International Classification of Ecosystem Services (CICES) framework (http://cices.eu/) is used to standardise ecosystem services. The CICES framework is an extension of the ES frameworks developed by the Millennium Ecossytem Assessment (MEA, 2005) and earlier studies (Costanza et al., 1997, De Groot et al., 2002, Daily et al., 1997). Apart from standardization for facilitating environmental accounting, a main objective of CICES is to provide a framework for more systematically approach the naming and describing ecosystem services. Compared to earlier Ecosystem Services frameworks the CICES classification has a higherarchical structure that accounts for issues of geographical scale and local contextualizations of specific ecosystem services classes. In addition, CICES also attempts to account for abiotic services and is continously developing, with classes and class types of ecosystem services being added as new applications emerge (EEA, 2016).

However, despite numerous studies on ecosystem services and natural capital, including biophysical assessments and efforts to model the flows of ecosystem services at specific geographical scales and contexts, knowledge gaps still exist (Lautenbach et al., 2013, Bull et al., 2016, Silvertown, 2015). An OPERAs report published in June 2014 identified and analysed the existing EU policy framework for ecosystem services and natural capital and concluded that it remains far from optimal (OPERAs D.4.1 IEEP, 2014). This is because the majority of existing instruments focuses on the regulation of ecosystems and thus lay primary importance on single ecosystem services, which does not address the full range of services that ecosystems provide (ibid.) nor does it necessarily take into account the underlying ecosystem function that give rise to ecosystem services. The relationship between ecosystem services and ecosystem functions requires more investigation in the future.

The role of governance research within the OPERAs project is to provide insights into particular aspects of governance for ecosystem services and how ecosystem services as a concept can be better included into policies and management strategies. A particular conceptual approach that we chose to use and that allows to better understand the complexities of the reality in which the ecosystem services concept will find application is through the construction of ideal types. In the following sections we provide a short overview of the current state of the art with regards to the governance of ecosystem services, then we introduce the concept of ideal types and how we conceive its usefulness and applicability for the governance of ecosystem services. We use two selected exemplars (see Table 1 for the list and short description of exemplars), and based on these we discuss how the notion of ideal types can serve as a conceptual tool in finding feasible approaches to operationalize the ecosystem services framework in diverse settings. The results of our collaboration with the two exemplars are used to inform this report.

1.1. Aims & Objective

This deliverable is part of the Work Package 'Knowledge' (WP3) in OPERAs. Based on available literature on governance of ecosystem services and data collected in two of the OPERAs exemplars, we constructed a typology of ideal types. Furthermore, we present findings from selected exemplars and to what extent these ideal types are already, or could be applied in the selected exemplars. We chose two of the twelve exemplars (Table 1), because they represent a variety of different ecosystem services, such as biodiversity habitat, carbon storage, flood protection, nursing ground for fisheries, aesthetic and landscape beauty and erosion protection. Moreover, since the ecosystem services that the exemplars focus on are mostly non-marketed, collective or public goods (Table 2), our analysis becomes more relevant compared to, for instance, marketed ecosystem services (e.g., timber, agricultural produce) for which existing regulations apply and market exchanges that manage the provision of these services already exist. Neverthless, our selection does not imply that marketed ecosystem services are necessarily better managed than non-marketed ecosystem services.

The two exemplars we selected for this milestone are the basis for an overview of existing and potential governance approaches, particularly in conditions where conventional applications of legal rights and / or public or private forms of regulations are blurred. This in turn allows us to apply the set of ideal types of governance modes with the objective to provide more targeted policy guidance.

Table 1 – A short description of the two selected OPERAs exemplars for this report

Exemplar	Short description of the exemplar
1) The Balearic Islands	Assess the co-beneficiary management of seagrass ecosystems for Blue Carbon, assessing magnitude of sinks, socioeconomic values, and management of tradeoffs- exemplar.
2) The Scottish Multi-scalar Exemplar (Focus on the Inner Forth area)	The Scottish exemplar works on four scales, with different spatial and thematic focus: building a science-policy-practice interface (ESCom), a national assessment of ecosystem services and policy, socio-cultural values of green space in peri-urban Edinburgh, and local benefits of coastal wetlands realignment.

2. Governance of ES

2.1. A brief history

Governance emerged as reaction to a previously quite narrow focus on government as the prime actor in shaping society. Governance implies the recognition that many more actors and structures are at play and that they interact in myriad ways. There is no universally accepted definition of governance, but there is wide agreement that governance today goes beyond regulation, public management and traditional hierarchical state activity. In addition to these traditional forms of political steering, governance emphasizes the use of novel instruments (such as voluntary and market-based approaches) and cooperative structures between state and non-state actors from various sectors of society (including the private sector, businesses and civil society). Most often governance implies certain degrees and forms of self-regulation and cooperation among different types of actors and coalitions (see Rhodes 1997 and Biermann 2007). Governance in its essence can be understood as the extent to which governmental and/or non-governmental actors (both private sector and civil society-related actors) are involved in governing. Moreover, governance equates to the totality of theoretical conceptions regarding governing (Arnouts et al., 2012, Kooiman, 2003).

According to Greiber and Schiele (2011) governance of ecosystem services encompasses, (i) the formulation of policies, referring to the processes and actors involved in the creation of policies, (ii) the implementation of policies and (iii) compliance mechanisms, related to how policies are controlled, monitored and also enforced. Herein, legislation and policies represent a crucial element of governance, since they provide visions, strategies, and plans for the management of common affairs. Thus, the governance of ES in that sense is also an agenda-setting, policy formulation and policy implementation issue on the same time. Moreover, Greiber and Schiele (2011) define governance of ecosystem services as the interaction of laws and other norms, institutions, and processes through which a society exercises powers and responsibilities to make and implement decisions affecting ecosystem services. Thus, governance of ecosystem services is the result of interplays of governmental, inter-governmental, and nongovernmental institutions, the private sector, and civil society based on rules established by statutory and customary law (Greiber and Schiele 2011).

Formal institutions and regimes to provide clear frameworks determining for instance where the power to make decisions is found and how responsibilities and accountability are distributed as well as compliance mechanisms. Therefore, governance benefits from a functioning legal framework which provides concrete and mandatory guidance on the management of ecosystem services. Nevertheless, governance should not be equated to government, since it is based on a much broader approach to governing, entailing more inclusion of relevant stakeholders and more deliberation.

However, considering man-made institutional set-ups and governance practices for ecosystem management with natural conditions and spatial distribution of ecosystems, it becomes apparent that ecosystem boundaries and political structures often do not match (Young, 2002). Ecosystems and their functions and services often span over geographical areas that fall into different political and

administrative boundaries and jurisdictions. Moreover, although several policy areas might be integrated to various degrees, the outcome leads to increasingly complex vertical and horizontal interactions across levels of biophysical, socio-economic and political structures (Young, 2013).

Public actors such as state or local authorities have a pronounced role in setting rules for the management and use of ecosystem services, as most ecosystem services and types of natural capital are characterized as public goods and services. This is important to bear in mind, because any actions to ensure protection or a more sustainable use of these lead to a benefit for those actors that protect, but also others who gain from the protection of ecosystems (Vatn et al. 2014). Moreover, because of the public good characteristic, governance of ecosystem services requires to look across scales that include many different social interests. These different social interests and the unequal distribution of power among the different interest groups is problematic since it risks to undermine the long term protection of ecosystems for and with relevant stakeholders, questioning the overall sustainability of these policies (see for example Paavola et al. 2009, Paavola and Hubacek 2013). Research on governance of ecosystem service is advancing, but is still largely minor compared to the overall field of ecosystem services research (Primmer et al., 2015). In order to close this gap, Primmer et al. (2015) have developed a framework for the the empirical analysis of the governance of ecosystem services. Their framework is based on the cascade model by (Potschin and Haines-Young, 2011) and they identify different modes of governance and incorporate them into the cascade model, which serves as a structure for empirical analysis of the cascade model with governance aspects. The framework proposed by Primmer et al. (2015) encompasses hierarchical top-down governance, scientific-technical governance, adaptive collaborative governance, and governing strategic behavior. Although potentially quite useful, we decided not to use the proposed framework for the empirical analysis but decided to use the notion of ideal types in this report, because it allows us to study the different and at times contradicting legislative and regulatory contexts that are at play in the exemplar contexts through a broader lens and independent of the cascade model.

2.2. Three modes of governance

Governance, following Rhodes (1997) understood as a the process by which society is governed or politically steered, tends to cluster over time into favoured sets of ideas, approaches and instruments used, so-called 'governance modes' (Kooiman, 1993, Lowndes and Skelcher, 1998, Howlett, 2009). Each mode has its distinct characteristics. The following brief presentation of the three modes (marketization, regulation and democratisation through deliberation) of governance is supposed to provide a short description of the three modes, but any governance arrangement will ultimately comprise a mix of elements of all three modes.

Marketization

The public sector increasingly adopts values and practices from the private sector in fields such as health, education and environmental management. This marketization trend is ubiquitous but particularly strong in transitionary economies with rapid industrialisation (Rigg, 2006). As a response

to the threat of global climate change a global carbon market was created and a new 'carbon economy'. The current global climate policy regime relies to a large extent on market mechanisms such as emissions trading, joint implementation and the Clean Development Mechanism. Regarding adaptation to climate change, insurance as an adaptation strategy represents a rapidly growing market where major financial players are increasingly active. Payments for ecosystem services (PES) is yet another emerging tool, applied from local to international levels, for the management of natural resources, such as biodiversity, water and soils (Pagiola et al., 2004). In the development debate, market integration is often described as a panacea (Sachs, 2005). While proponents of marketization argue that markets are most effective for dealing with problems, opponents fear that this will compromise values related to democracy, citizenship (Eikenberry and Kluver, 2004) and equity (Rigg, 2006). Thus, in the context of the research agenda on sustainability challenges, marketization has to be scrutinised for its effectiveness and its impact on social justice.

Regulation

There are profound challenges regarding legal regulations of sustainability. While environmental problems are often trans-boundary and trans-jurisdiction, much regulation is based on national law. New forms of regulative bodies transcending the nation state are therefore needed. Since there is no legal bearer of a right belonging to future generations, contemporary law is challenged by the intergenerational approach to sustainability. We therefore need more emphasis both on regulatory techniques and ethical principles (Gunningham et al., 2003). One initiative in this direction is seen in climate politics with the concept of the 'ensuring state' that serves as catalyst, facilitator and provider of guarantees in relation to both citizens and other states; this would imply a new form of strong state (Giddens, 2009). The global research programme Earth System Governance aims to contribute to new forms of governance at the planetary (and local) level (Biermann et al., 2009). A suggested task here is to critically rethink contemporary regulative processes from a normative perspective.

Democratisation through deliberation

The strong deliberative turn in democratic theory during recent decades speaks to an emerging concern with the distance between the interests and motives of citizens and the decisions made in their name (Smith, 2003). A growing scholarship today questions liberal democratic institutions by pointing at the lack of voice of citizens and the poor representation of ecological values in decision-making processes (Dryzek, 1997, Eckersly, 2004). Deliberative democratic theory has evolved as a response to this perceived weakness of liberal democracy. It seeks both to democratise and to 'green' policy discourses by increasing the opportunities for citizens to engage in decisions that affect their lives and surrounding environment (Dobson, 2003). The deliberative project also extends to the international arena and has been forwarded as a strategy that can bridge the democracy deficit in governance arrangements beyond the state (Nanz and Steffek, 2005) and foster a trans-national green public sphere (Dryzek, 1997). Research in this sub-theme should seek to examine how 'democratisation through deliberation' plays out in the environmental domain. We are particularly concerned with the potential synergies and tensions between the substantive and procedural aspects built into the deliberative project. As Goodin (1992) famously claimed, "(t)o advocate democracy is to advocate procedures, to advocate

environmentalism is to advocate substantive outcomes." Hence, how and to what extent can a deliberative model of democracy represent a pathway towards sustainability?

2.3. Instruments for the governance of ES

In this section we present a short typology of ecosystem services in table 2, where we categoriza ecosystem goods and services according to their economic characteristics, that is their level of rivalry (i.e., the use by one person physically precluding others from using it) and whether or not laws are in place that regulate access.

Currently, there are three main approaches that are used in the governance of ecosystem services. The first group can be characterized as marketization/commercialization. The basic idea behind marketization or commercialization is to let demand steer management practices that result in desired outcomes, i.e., the production and delivery of certain ecosystem services. For example, in table 4, those ecosystem goods and services for which laws are in place that prohibit access can be governed by using market approaches. For those ecosystem services that fall into Box 1, markets often do already exist or could potentially be created, for instance in the form of some direct payments from users to owners of the ecosystem service (i.e., private fishing lakes, hunting licenses) or as Payments for Ecosystem Services (PES) schemes.This is primarily the case for already marketed goods (agricultural products) and to some extent also for collective goods. For those ecosystem services that fall into Box 2, which are few, market approaches need to be devised artificially (artificial scarcity to steer demand) for example by governments through private-social or public- private partnerships.

Table 2 - A typology of ecosystem services and their economic characteristics

		Does use by one person physically preclude use by others?	
		Yes – Rival	**No – non-rival**
Do laws prohibit access to these services?	**Yes – excludable**	**BOX 1** **Market goods** – agricultural products (Cultivated crops, reared animals, in-situ aquaculture), to some extent carbon sequestration and storage **Non marketed goods** – wild plants and animals, fisheries in Marine Protected areas and trawling fisheries in the entire ecosystem ***Mostly provisioning services***	**BOX 2** **Collective goods / club goods** – (Artificial scarcity to steer demand); access to community owned forest or park for recreation for instance restricted access of visitors to enjoy national parks (Cabrera Archipelago NP in Mallorca) ***Many cultural ecosystem services (recreation, etc.)***
	No – non-excludable	**BOX 3** **Common property resource** – wild fish stocks, timber from unprotected forests, aquifers, fisheries other than trawling outside Marine Protected Areas ***Mostly provisioning services***	**BOX 4** **Pure public goods** – climate regulation, pollination and seed dispersal (by wild populations), many CES (aesthetic, spiritual), carbon sequestration, sand production, nutrient removal ***Regulating and Maintenance services plus cultural ecosystem services (spiritual, symbolic and other interactions with land - / seascape)***

A second group of ecosystem services is left largely unregulated, it is those for which currently no laws are in place that prohibit access, or if there are laws they are often not sufficiently enforced. These ecosystem services are however rival and their use by someone limits or even prevents someone else from using them (Box 3 in table 2). In theory, it is also possible to create access rules or rules over who can use these services and how. In practice, however, this might not always be feasible, or might lead to undesired consequences and inequities over who gets to decide and who gets to benefit from these ecosystem services in the future.

The third group of ecosystem goods and services falls into the category for which neither laws are in place that restrict access nor is the use by one person preclusive for the use by others (non-rival, Box 4 in table 2). For this group, most feasible and equitable form of governance is through democratization, characterized through dialogues and jointly set targets by all concerned and potentially affected stakeholders or citizens (Weber's notion of value or emotional rationality). Nonetheless, governance attempts to regulate currently unregulated ecosystem service. The most prominent example is carbon sequestration and storage from anthropogenic greenhouse gas sources. For carbon, artificial markets have already been created in some countries and regions (for example the EU ETS carbon trading

scheme), but they depend to a large extend on political factors and state interests, as well as negotiations between private sector interests and states in their inception and functioning (Vatn, 2014).

In the past, mainly command and control mechanisms, such as legal protection and public provisioning, were used to manage social-ecological systems. However, the focus on command and control mechanisms was increasingly criticised due to unexpected social and environmental problems caused by the attempts to control highly complex and nonlinear natural systems (Holling and Meffe 1996, Folke et al. 2005). The turn away from mostly command and control mechanisms led to a range of governance instruments for ecosystem service that are already applied in various settings. In table 2 we provide a short overview of instruments of environmental policy, grouped into regulatory and legal rules, information and motivational instruments and economic instruments.

Increasing attention is drawn to a variety of economic instruments, which can comprise both market and non-market types. Trade characterized through market types represents a particular set of governance structures that already exist or that are on the way of being implemented to manage and govern ecosystem services. Payments for ecosystem services (PES) are one prominent example for this form of trade-based governance, although they often operate in the grey zone between market and non-market types (Vatn 2014). On the other hand there are a variety of non-market based structures, such as laws and regulations at different administrative and political levels. In the European Union the aforementioned Biodiversity Strategy to 2020 and the Birds and Habitats Directive are examples of regulatory frameworks established at a supra-national level. Furthermore, non-market-based approaches include for example subsidy reform, land use activities and different policy instruments, certification and labelling initiatives. Within the EU, there are attempts to implement biodiversity offset markets, which is supposed to function like a complete market with intermediaries. Herein, public bodies are involved as regulators who define goals, control trades and performances (Vatn 2014). All policy instruments, either market or non-market, have a certain command element, because rights need to be defined. If a landowner is participating for example in a market scheme (which by definition is voluntary), or is affected by certain regulations that delimit her ability and ways to use land, property rights are a pre-requisite (Vatn et al. 2014).

Table 3 - Environmental policy instruments (based on Vatn et al. 2014; Ring and Schröter-Schlaack 2011)

Policy instruments	
Regulatory and legal rules (often called command and control)	**Public provisioning:** - For instance rules regarding use of resources or the protection of public lands - Rules regarding the assessment of impacts (e.g., Environmental Impact Assessments, Strategic Environmental Assessments, Appropriate Assessments) - Permits - Standard-setting and zoning or planning **Legal protection:** - Prohibition & restriction of damaging activities - Mandated solutions - Protection – e.g., National Parks, Nature Reserves, Natural Heritage Sites
Informational and communicative instruments	**Shift individual or community preference functions** - Technical information - Normative - Education - Development of skills
Economic instruments	**Pure public instruments:** - Taxes, charges and fees (pricing environmentally damaging behavior) - Subsidies for certain land uses - Fiscal transfers
	Market instruments: - Payments based on contracts (reward conservation enhancing behavior) - Public auctions - Cap-and-trade systems - No net loss / biodiversity offsetting

We decided to represent policy instrument in table 2 in three categories, which is one commonly accepted way of doing it. With regards to EU instruments and their relevance to ES and NC, a different structure can be explored. OPERAs partners in WP4 assess policy instruments from the starting point of practical decision making, identifying and categorising them as 1) information instruments (i.e. instruments that help to provide relevant information to the decision- making process), 2) decision-support instruments (i.e. instruments used as aids to the decision-making such as EIAs) and 3)

implementation instruments (e.g. regulations, rules and market-based instruments integrating ES/NC into the resource use in practice) (OPERAs D.4.1 IEEP, 2014). In the further work of OPERAs, we will continue to explore different forms of categorisation of policy instruments and governance approaches relevant for ecosystem services and natural capital.

3. The concept of ideal types

In its pure form, an ideal type is a simplified but exaggerated conceptual tool that is used to model reality. Ideal types are employed in social science to illustrate a concept. Initially, ideal types originate from Max Weber's argument that no scientific system is ever capable of reproducing all concrete reality, nor can any conceptual apparatus ever do full justice to the infinite diversity of particular phenomena (Weber 1904/1949). Therefore, for an investigator or researcher an ideal type is an analytical construct that serves as a measuring rod to ascertain similarities as well as deviations in concrete cases. Importantly, an ideal type is not meant to refer to moral ideals or a perfect reality, it rather serves an accentuation of typical courses of conduct (for example in collective actions of individuals in society). Moreover, ideal types do not correspond to concrete reality but always move at least one step away from such a reality. Because it is constructed using certain elements of reality that form a logically precise and coherent whole, ideal types cannot be found as such in reality. Yet, they are useful insofar as they can provide a basic method for a comparative study of certain aspects of reality, which share a commonality, for example the concept of ecosystem services.

Ideal types enable the constructions of hypotheses linking them with the conditions that brought the phenomenon or event into prominence, or with consequences that follow from its emergence. As Julien Freund (1968: 69) puts it, "Being unreal, the ideal type has the merit of offering us a conceptual device with which we can measure real development and clarify the most important elements of empirical reality." Ideal types should be seen as thought experiments based on empirical observations that help us create logically coherent and objectively feasible configurations of social relations, and thereby guide policy making and governance (Jessop, 2002).

3.1. Why ideal types for the governance of Ecosystem Services and Natural Capital

Ecosystems perform functions independent of people, but anthropogenic activities have strong impacts on how ecosystems function. A wide range of ecosystem functions (for example nutrient or carbon cycling) is particularly useful for people, because it provides the foundation to produce food and other agricultural commodities, for flood regulation, carbon storage in biomass and soil or water purification for human consumption. Yet, despite the numerous services and benefits that people and ultimately society derives on a daily basis from ecosystems, their governance is often complex and challenging. Governance of ecosystems is characterized by processes of negotiation between different groups of stakeholders, for example farmers, governmental institutions and authorities, who have diverging and at time conflicting interests and different levels of power.

Dietz et al. (2003) stated that environmental governance depends on good and trustworthy information about stocks, flows, and processes within the resource systems being governed, as well as about the human-environment interactions affecting those systems. However, when scales are highly aggregated, information may ignore or average out local information that is important in identifying future

problems and developing solutions. Therefore, effective governance requires not only factual information about the state of the environment and human actions, but also information about uncertainty and values (Dietz et al., 2003). In addition, the scientific understanding of coupled human-biophysical systems is always uncertain, because of inherent unpredictability in the systems and because the science is incomplete (Wilson 2002). Against general perceptions, science does not fully understand ecological complexities, trade-offs and feedback loops, nor does it grasp the full spectrum of social factors that influence human preferences, decision-making, values and behavior, which bear on the governance of natural resources and ecosystem services (Wilson 2002). Thus, we suggest that by using the notion of ideal types, we can construct and model the complexity of reality in order to provide better ways to govern ecosystem services.

Ideal types are simplified conceptual tools that can be used to better describe and understand certain phenomena in society. One can then approach these from a scientific lens. From that perspective, ideal types are useful in reflecting about appropriate governance tools for ecosystem services and natural capital. Nonetheless, ideal types should by no means be understood or seen as representing an 'ideal' or 'perfect' solution, since they are, as previously stated, simplified and do not capture the entire complexity of reality. The main premise of the construction of ideal types is to come up and provide concrete and policy relevant guidance on how to govern ES, anchored in specific local contexts and social-ecological conditions.

There have been a few attempts to construct ideal types for the governance of ecosystem services. For instance, Arnouts et al. (2012) present a framework, based on Kooiman's (2003) governance conception, of four ideal type governance modes that are operationalized into four ideal-type governance arrangements (Table 3). They apply their framework to a case study of the rise of Dutch Nature policy. The ideal-type governance arrangements are a continuum from government-centered governance (i.e., hierarchical to closed co-governance) to non-state governance (open co-governance to self governance).

Table 4 - Overview of four ideal typical governance arrangements, Source: Arnouts et al. 2012

Ideal-typical governance arrangements				
	Hierarchical	**Closed co-governance**	**Open co-governance**	**Self governance**
Actors	Mainly governmental actors	Selected mixed group of actors	Large mixed group of actors	Mainly non-governmental actors
Power	With government	Pooled	Diffused	With non-government
Rules	Governmental coercion	Restricted cooperation	Flexible collaboration	Non-governmental forerunning

3.2. Diverging theories of social systems – implications for the governance of ES

Ecosystem services are a human-centered concept and have emerged out of the necessity to show the value of those ecosystem functions that lead to human benefits, both directly yet also indirectly. This instrumentalisation of nature through ecosystem services and natural capital and attempts for their valuation has received large criticism (Martinez-Alier et al., 1998, O'Neill et al., 2008, Norgaard, 2010, Melathopoulos and Stoner, 2015, Silvertown, 2015). Nevertheless, despite being a normative anthropocentric concepts, ecosystem services and natural capital are also considered useful in order to account for nature in economic and political decision-making (Gomez-Baggethun and De Groot, 2010, Bull et al., 2016, Schröter et al., 2014).

Concepts such as ecosystem services have helped to bridge the social and natural spheres, leading to research that is increasingly transdisciplinary (Lundy and Wade, 2011, Siew and Döll, 2012). Most approaches to address the loss of ecosystem services are based on economics and rational behavior, which might create more problems than solutions (Kosoy and Corbera, 2010, Van Hecken and Bastiaensen, 2010, McAfee, 2012). A major reason for this is related to the existence of two different perspectives in the social and natural sciences. An important source of incommensurability between the social sciences and most natural sciences interested in social processes such as environmental degradation and pollution is how society is understood and represented. We can identify two types of approaches to understand society, resembling what in sociology is called consensus theory and conflict theory.

According to consensus theory, shared norms and values are the foundation of a stable harmonious society in which social change is slow and orderly. For example, when using the concept of coupled social ecological systems, resilience can be seen as the equivalent of stability and harmony or the good norm (Olsson et al., 2015). In contrast, conflict theories emphasise conflicting interests between groups in society meaning that social order is maintained by (material or discursive) manipulation and control by dominant and powerful groups, and that transformational change can develop from the tensions between these groups and the redistribution of power. According to conflict theory, institutions are shaped by existing conflicts, power (im)balances and social stratifications in society. This has implications on how governance mechanisms are understood differently if we rely on consensus theories or conflict theories.

Another important dividing line in the social sciences is also how human behaviour is understood, i.e. on what reasons we, that is, policy-makers, corporations and individuals make decisions. In rational choice theory, the foundation of neoclassical economic theory, individuals make their decisions based on maximising their own utility. The simple assumption of rational choice provides the basis for modelling of the economy as a self-organising system. It also provides a scientific justification for the current proliferation of market-based instruments for ecosystem management as epitomized by 'The Economics of Ecosystems and Biodiversity' initiative (TEEB) aiming to 'help decision-makers recognize,

demonstrate and capture the values of ecosystem services and biodiversity' (Kumar, 2010, Brown, 2014).

Rational choice theory is, however, deeply contested in the social sciences. Other and more elaborated theories for explaining social behavior have been formulated in sociology, such as various institutional theories and symbolic interactionism. Symbolic interactionism, in sharp contrast to rational choice theory, is based on three foundational assumptions (Blumer, 1986):

- Humans act toward things [or the natural environment] on the basis of the meanings they ascribe to those things (i.e. things have no universal value in themselves)
- The meaning of such things is derived from, or arises out of, the social interaction that one has with others and the society (i.e. decisions are primarily relational rather than individual),
- These meanings are handled in, and modified through, an interpretative process used by the person in dealing with the things s/he encounters.

Thus, when considering the socio-political dimension of ecosystem services and the social context in which ecosystem services are created, valued and used, it is crucial to approach these with sociological theories. There is, however, an increasing trend in governance of ecosystem services research towards acknowledging this divide and accept that rational choice theory, the underlying assumption guiding the proliferation of market-based mechanisms, alone is not sufficient to understand and explain why individuals and groups act in certain ways. In particular, with regards to valuation of ecosystem services for instance, sociology and behavioral economics are increasingly used to understand potentially negative effects of governance of ecosystem services through market-based approaches (Rode et al., 2015, d'Adda, 2011).

4. Key questions for constructing ideal types

We developed a set of generic questions that were sent out to selected OPERAs exemplars in the course of 2014. These questions served as a starting point for the enquiry and the construction of the ideal types for selected ecosystem services (Table 5) and the subsequent application to the two exemplars we chose. In table 5, we list the key questions and the way we envision governance modes for selected ecosystem services. It is important to note here that the key questions are only intended to provide a basis to better understand and analyze the current state of ecosystem service governance. By no means do we infer that property or user rights are always needed nor do we suggest that establishing these should be a goal.

Table 5 - Ideal types and selected ecosystem services

Key questions	Example of Ecosystem services			
	Pollination and seed dispersal	**Flood protection (in river catchments and watersheds)**	**Water purification by ecosystems**	**Global Climate Regulation (Forest Carbon Sequestration and Storage)**
Are the property rights arrangements clear?	Unclear	Clear	Clear	Above ground: clear Below ground: unclear
Are the user rights arrangements clear?	Unclear	Clear	Clear	Clear
Do we understand the science?	Low	Very high	High	Medium
Are the boundaries of the systems defined / definable?	Fuzzy boundaries – species dependent ranges, highly mobile and numerous overlaps	Clear boundaries (catchment / watershed)	Surface: clear; Sub-surface: unclear	Clear boundaries (but leakage is a risk)
Are there temporal inertia and lags?	Yes	No	Yes	Yes (permanence issue), multi-generations
Can the stakeholders be defined?	Partly – Low definability	Yes – high definability	Yes – high definability	Partly - Low definability (multi-generations)
Are power relations among the stakeholders clear?	Unclear (Polarization)	Clear (Consensus)	Unclear (Polarization)	Unclear (Polarization)
Production / distribution rules	No distribution rules – public good	Marketed good / non-marketed good - Market-distribution	No distribution rules - Non-marketed good	No distribution rules - Collective good / common property resource

Ideal typical mode of governance (based on Arnouts et al. 2012)	Open co-governance	Closed co-governance	Hierarchical / Closed co-governance	Open co-governance / Self-governance
Examples for ideal typical mode of governance & policy instruments	Regulatory and legal rules incorporating the precautionary principle, informtation and communicative instruments	Payment for Ecosystem Services Schemes (PES)	Protected areas	Land use and land management rules, voluntary schemes, carbon markets, burden sharing agreements and emission targets

We use the key questions posed in table 3 to construct the ideal types. They are meant to indicate the challenges that continue to exist when contemplating which governance approaches are the most appropriate given the current conditions. Therefore, the concept of ideal types is useful as it allows disregarding some specificity, but focuses on a simple yet comparable form of understanding ecosystem services situated in specific social and institutional settings. For instance, policies for ecosystem service management and regulation need formal institutions to provide clear legislative frameworks. These frameworks then determine, for instance, where the power to make decisions is found and how responsibilities and accountability are distributed as well as what compliance mechanisms are put in place.

The OPERAs projects and most of its exemplars are situated in a European context. This has implications for governance approaches for ecosystem services. In times of increasing Europeanization countries within the European Union cede parts of regulatory processes to the EU. In terms of environmental regulations and legislation, this manifests through a number of EU directives and EU regulations, such as the Birds and Habitat directive, which have to be directly implemented by member states. EU directives have a greater importance than national regulations, the latter having to be adjusted to EU directives which supercede national legislation. The unique situation with the EU has implications on the governance of natural capital and ecosystem services in the EU member states, national regulations that implement EU Regulations cannot contradict each other. National Regulations that are drafted to implement EU Directives however may be slightly contradictory, but nevertheless national legislation must follow the spirit of intent of the Directives each other. Here again, the ideal type concept can be used to overcome this issue, by focusing on the most important elements for the governance of ecosystem services. We have identified these elements in the key questions.

An additional way of approaching the governance of ecosystem services comes from examples of transnational governance. Here, regulatory standard setting is a process that goes beyond the state as the entity responsible for setting standards of production (Abbott and Snidal, 2009). For marketed agricultural products, fisheries and forest products, but also for industrial processes that require natural resources inputs, certification represents an approach that is increasingly used in order to set and promote standards beyond those that are legally demanded. Actors involved in setting standards and participating in their oversight include different groups from private sector institutions and firms, public entities and states, and NGOs. According to Abbott and Snidal (2009) states are not obsolete as a

regulator, but their role as agenda setting actors has changed substantially and increasingly moves away from direct regulation towards the support of regulatory standard setting schemes.

Nowadays, increasing attention is drawn to standard setting and a variety of other economic instruments, which comprise both market and non-market types. Economic instruments, either market or non-market, still have a certain command element, because rights need to be defined. Trade characterized through market types represents a particular set of governance structures that already exist or that are on the way of being implemented to manage and govern ecosystem services. Payments for ecosystem services (PES) are one prominent example for this form of trade-based governance, although they often operate in the grey zone between market and non-market types (Vatn 2014). On the other hand there are a variety of non-market based structures, such as laws and regulations at different administrative and political levels. In the European Union the aforementioned Biodiversity Strategy to 2020 and the Birds and Habitats Directive are examples of regulatory frameworks established at a supra-national level. Furthermore, non-market-based approaches include for example subsidy reform, land use activities and different policy instruments, certification and labeling initiatives. Within the EU, there are attempts to implement biodiversity-offset markets, which are supposed to function like a complete market with intermediaries. Herein, public bodies are involved as regulators who define goals, control trades and performances (Vatn 2014).

5. Explaining the key questions

5.1. Intersecting policies at EU, national and local level

The ES/NC concept strengthens the thinking in systems—not only in terms of ecosystem processes and functions, but also with regard to social and political systems—and emphasizes the linkages between ecological and human systems (Costanza et al., 1997) However, considering governance practices in the various relevant policy areas discussed below, it becomes obvious that ecosystem boundaries and political structures often do not match (Young, 2002). Moreover, the various policy areas are integrated to various degrees, leading to increasingly complex vertical and horizontal interactions across levels of biophysical, socio-economic and political structures (Young, 2013).

Agricultural policy: Agriculture policy is the most integrated of all EU policies (Wallace et al., 2010). Even though the share of agricultural policy expenditures on the total EU expenditure has decreased from almost 75% in 1985 to around 43% in 2013, agricultural policy is still the biggest EU budgetary item (Commission, 2013). In 2013, the budget for the two pillars of the Common Agricultural Policy (CAP), farming subsidies and contributions to promote rural development, is 57.2 bn Euros. How agricultural policy reforms are designed is absolutely critical for the support and maintaining of many critical ecosystem functions and services (Plieninger et al., 2012) as they affect land use patterns and management practices for a substantial share of land in the European Union.

Cohesion policy: Regional aid is the next biggest item in the EU budget (around 47bn euros for 2013). The EU cohesion policy has the potential to be a key tool in the implementation of the EU's growth strategy 'Europe 2020', which aims to better gear towards sustainable development by addressing a wide range of economic, environmental and social objectives (IEEP, 2011). The scope of cohesion policy is regional development. This leads to a range of impact on ES, such as indirect impact on ES use by promoting infra structure, growth and jobs.

Biodiversity policy: Biodiversity policy has been defined as one of the main policies to maintain ecosystem functions and services, both at the global and European level. Very challenging remains the translation of biodiversity protection goals and strategies into specific policy measures and to integrate biodiversity concerns into all EU and national policy sectors (European Commission, 2011).

Water policy: The EU water framework directive integrates ecosystem-based objectives and planning processes at the level of river basins into water resource management across Europe (European Parliament and Coucil, 2000). The scope of the water framework directive is river basins in the EU, and the impacts vary, such as reducing pollution of freshwater thereby increasing habitat quality for numerous species and other ES.

Fishery and maritime policy: A common European fisheries policy was established in 1982; the European Commission developed a Maritime Strategy Directive and an Integrated Maritime Policy in 2005 and 2007, respectively (van Hoof and van Tatenhove, 2009). Nevertheless, the common fisheries

policy of the European Union has often critiqued for not living up to expectations and for not achieving the protection of European fish stocks, thus undermining many ES (Daw and Gray, 2005, Khalilian et al., 2010).

Nature and landscape protection: Policies for protected areas are mainly a national domain. The European Landscape Convention under the Council of Europe (2000) aims at promoting the protection, management and planning of European landscapes and seeks a better coordination of related activities.

5.2. Jurisdictions involved and affected

The implementation of EU policies as well as the allocation of authority and competences in predominantly national (or subnational) policy domains may vary greatly from one country or region to another. These differences can be expected to have significant consequences for various ecosystem functions and services and how they are governed across jurisdictions. Questions regarding jurisdiction relate to the following:

- Which jurisdiction(s) applies in the exemplar cases?
- Are there legal rules that aim specifically at ecosystem functions or ecosystem services?
- Are these jurisdictions overlapping, contradicting, complimentary?

5.3. Property Rights arrangements

Property rights are embedded in social, political, cultural and economic contexts and have an important effect on how humans interact with their environment (Ostrom 1990, Hanna and Munasinghe 1995). Property rights regimes influence the use of environmental resources, a fact that has long been well established, if not well practiced. Essentially, property rights consist of bundles of entitlements through which rights and duties in the use of natural resources are defined, while property rules refer to the rules under which those rights and duties are exercised (Bromley 1991).

A property right is the exclusive authority to determine how a resource is used, whether that resource is owned by government or by individuals. Society approves the uses selected by the holder of the property right with governmental administered force and with social ostracism. If the resource is owned by the government, the agent who determines its use has to operate under a set of rules (Alchian 2008).

Thus, property rights arrangements are important in order to determine what role land-owners, land-users play in the maintenance and provision of ecosystem services, and to what extent their actions are influenced or regulated by laws and regulations (either regional, national or European). Considering the example of ecosystem functions that give rise to socially desired ecosystem services, it is crucial to determine what kind of property rights exist and through which institutional settings these are affected or transformed, as well as the implications this might have on the ecosystem functions itself and on the ecosystem services that are the result of these functions. To elicit issues regarding property rights, we

ask several questions to exemplars in order to understand the contextual aspects under which property rights play a role for the governance of ecosystem services. In the list sent to the different OPERAs exemplars we ask:

1. **What can be owned?**

For example, if a person owns a piece of land does it also imply that she owns the water flowing through the property, or the insects pollinating plants, or minerals (including soil carbon) that might be found in the ground?

2. **Who can own?**

In most European Union countries, land can be owned by private landowners and companies, without limitations on the size of the land property. With regards to marine environments however, ownership is quite different and marine ecosystems cannot be owned privately. With sea-level rise, particularly ownership of land in low-lying coastal areas is bound to be transformed in one way or another. In one of the exemplars, we study the possibility for managed realignment land that is in the process transformed to tidal habitats, partly covered by water. This has implications on ownership structures. Moreover, depending on ownership structures, stakeholder vary and policy instruments that are implemented will change.

3. **What can be done with it?**

Ownership does not automatically imply unrestricted user-rights, but such restrictions vary from place to place and also with type of property.

4. **How can ownership be maintained?**

In some cases the maintenances of ownership is straightforward, for example in the case of private ownership of real estate that is registered by an authority. But there are many examples where ownership is fuzzy and contested and where the concept of entitlements by Amartya Sen could be applied in some cases (Sen 1981, Leach et al. 1999).

5. **How can ownership be transferred?**

Buying and selling in the market is perhaps the most common way of transferring ownership, but definitely not the only way. Can ownership for example be transferred to future generations? What happens to user-rights when ownership is transferred? Can future land-use be conditioned upon transfer of ownership, if so, for how long and even for future transfers of ownership?

The points we raised above elucidate the importance of a critical discussion regarding property rights over ecosystem functions and services. This is particularly so when the object or entity in question is itself neither well-defined, fall within several administrative boundaries and are mobile agents (seed

dispersers, pollinators, etc.). More so, even from an ethical and normative perspective this issue requires a critical analysis and informed debate before framing ecosystem functions and ecosystem services as something that can be owned by a private person or, for that matter, a corporation.
To highlight just a few concerns, we need to better understand:

- What the existing and potential future policy integration needs are,
- What relevant and important cross-jurisdiction issues arise in the management of ecosystem services at different scales and levels,
- How property rights arrangements affect the management of ecosystem services, and
- What role property right play for ecosystem services?

For example, while property rights (PR) arrangements are highly contextual and have a decisive impact on resource distribution, governance arrangements, economic performance and (in)equality in local settings, they are also important at national, regional and global scales. In the European Union, agricultural and forest land can be privately owned, thus targeting land owners as resource users and managers is one way for services such as carbon storage in forest lands etc. But some ecosystem services, for instance the regulating and maintenance service pollination, are a mobile-agent-based ecosystem service (Kremen et al. 2007) and as such geographical boundaries become obsolete. Another such example is soil carbon, which is an example for the problematic assignment of property rights. Soil contain large amounts of carbon (Schils et al. 2008), and the question that arises in this context is who can own the ecosystem service of soil carbon sequestration and storage? A similar problematique surfaces in the governance of managing a mobile-agent based ecosystem service, such as pollination. These are just two examples of highlighting the issues with regards to governance of two specific ecosystem services that requires further analysis.

Rather, instead of property rights, the notion of entitlements can be more appropriate (Sen, 1981). Entitlements are a different conceptualization of ownership relations based in certain rules of legitimacy. Sen further develops four entitlement relations that are accepted in a private ownership market economy such as (1) trade-based, (2) production-based, (3) own-labor and (4) inheritance and transfer entitlements (see Sen 1981).

5.4. User rights arrangements

Are there user rights that are different from property rights. For instance, de-facto rights that refer to rights that exist in practice but that are not necessarily ordained by law or that are not officially established. De-jure rights refer to established laws that determine rights. This distinction is important for understanding how user rights are maintained and how they can be transferred. Many of the five key questions under property rights are also applicably to user rights.

5.5. Understand the science

In the past decades science has made tremendous advances and there are an increasing number of studies on the complexity of ecological systems and the intricate interactions between social and ecological systems. Social ecological systems theory has become an important field of interdisciplinary study, particulalry relevant for the governance of ecosystems and ecosystem services (Berkes et al., 2003, Ostrom, 2009, Fischer et al., 2015). Nevertheless, science is not complete and the complex interactions of some ecological systems and their response to human pressures is not always entirely understood, and proxies that are being used may in fact confound our understanding of the ecological complexities (Norgaard, 2010). Pollination for instance is an important ecosystem service. Numerous studies have investigated pollination services, from the local to the landscape scale, but some interaction between human pressures and land-use change on pollination are not well undersood (Manley et al., 2015, Schulp et al., 2014, Bennett et al., 2014, Lundin et al., 2013). For instance, we only recently came to understand how for instance bees and other pollinators are affected by neonicotinoids (Smith et al., 2013), which sparked debates on the use of science and the application of precautionary principle in the European Union to the effect that the substances were banned in some countries but not in others (Cressey, 2013, Van der Sluijs et al., 2013, Woodcock et al., 2016).

5.6. Defined boundaries of the systems

The definability and spatial extent of ecosystem services and their underlying ecosystem functioning is an important component to determine the spatial scale for the governance of these services and also to determine the direct users and stakeholders. For example, in the context of water provision or flood regulation, the definability of the system and its spatial limits are exact, namely a watershed under the EU Water Framework Directive (European Parliament and Coucil, 2000). Yet, in the case of pollination services, the limits and spatial extent of the ecosystem functioning and services are hard to determine.

5.7. Temporal inertia and lags

Are there lags and inertia in the relationship between ecosystem functioning and the ecosystem services? Lags and inertia increases the risk of undermining the ecosystem services. For example soils can degraded for quite a long time without reducing the ecosystem service if nutrients are replaced through other means, such as chemical fertilizer, than the natural nutrient cycling. But ultimately there will be a reduction in the services provided by the soil which may take very long time to restore. Understanding temporal inertia and lags in ecosystems is a fundamental aspect for the governance of ecosystem services, because it means that ecosystems and the services are not commensurable, neither in their ecological value nor their socio-cultural value. This has implications for policies that promote the idea of biodiversity offsetting or no-net loss (Robertson, 2004, Gardner et al., 2013).

5.8. Identifying stakeholders

What are the most important stakeholders of different categories (state, private, civil society)? Are people who live in and around the exemplar areas able to partake in shaping decisions regarding their present and future management?

5.9. Power relations among the stakeholders

What are the most important power relationships among the stakeholders? This question relates and attempts to elucidate for instance the structural powers at play, where are the legislative powers place and how does normative power shape the way society is responding to proposed changes.

5.10. Production / distribution rules

What characteristics does the ecosystem service exhibit, e.g., is the service in question rival in use or can people be excluded / prohibited from using it or not? In table 2 we give a short overview of the classification of different ecosystem services with examples that we draw upon.

6. Applying ideal types to the two exemplars

In this section we present the two selected exemplars in more detail and provide data and information on the questions that we use to identify ideal types of governance modes for the ecosystem services in each exemplar setting.

6.1. The Balearic Islands

Within the suite of OPERAs exemplars, the Balearic Seagrass meadows (*Posidonia oceanica*) represent a unique case, since it is the only marine ecosystem. The exemplar's main goal is to assess the co-beneficiary management of seagrass ecosystems for Blue Carbon, assessing magnitude of carbon sinks, the socio-economic values of the seagrass meadows, and the management of tradeoffs.

Figure 2 - Posidonia oceanica meadows around the Balearic Islands, Photo by Juan Cuetos (Oceana)

Posidonia oceanica meadows are a marine ecosystem, which has several important differentiating traits. **First**, the suite of EU policies and directives that apply is different compared to the land-based ecosystems. **Second**, ownership structures are distinct, seagrass meadows as part of the coastal fringe, and therefore it cannot be owned by private entities, but are entirely owned by the government. **Third**, with regards to the ecosystem services these seagrass meadows provide, locals and visitors differently appreciate a direct benefit and use of the ecosystem. According to interviews we conducted with policy and decision-makers as well as experts, awareness abou the benefits of Posidonia oceanica is very low in the Balearic context. The interviewees stated that alhtough local stakeholders with a very close relation to the marine environment recognize a direct benefit of seagrasses, a majority of people who live on or visit the Balearic Islands are not aware of the benefits Posidonia oceanica provide (see Figure 3 below). The main reason for this is the very underwater nature of the seagrass meadows. Being a marine ecosystem, most people do not see the meadows with their own eyes, nor will they be able to establish a personal link with the ecosystem and its services (Kurani 2015). This has direct implications for the governance of Posidonia and the use of socio-cultural valuation as a method to derive people's preferences.

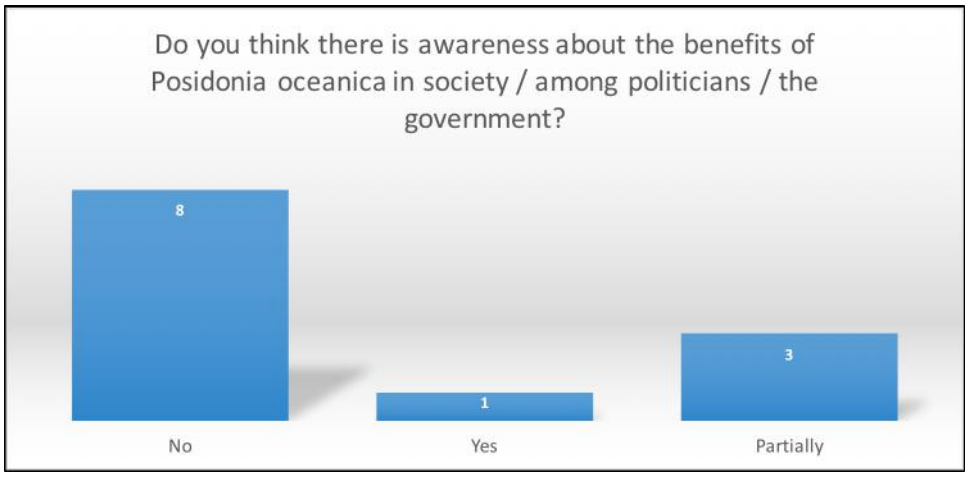

Figure 3 - Awareness about the benefits of Posidonia oceanica

Intersecting policies at EU, national and local level

The Posidonia oceanica meadows are included in the Spanish national and EU level regulations. The EU Habitat Directive (92/43/EEC) recognizes Posidonia oceanica as a priority habitat for conservation. At the Spanish national level various laws protect Posidonia oceanica (National law 4/1989 on the Conservation of Natural Areas, Flora and Wildlife; Royal Decree of December 7, 1995 (BOE 310, 28/12/1995) and establish necessary measures for their conservation. The Royal Decree 139/2011, 4th of February, includes Posidonia oceanica in the List of Wildlife and Flora with special protection status. This means that under Spanish law, Posidonia oceanica is a protected species and any removal of the plant, alive or dead, is prohibited and punishable (Royal Decree 139/2011).

Moreover, because of the detrimental effects of fishing by bottom trawling, the Council Regulation (EC) No. 1626/94 specifies that bottom trawling be expressly forbidden on seagrass meadows. In addition, In the Balearic Islands regional law prohibits any type of trawling above 50 meters depth (BOE 169, 16/07/1962). Because Posidonia oceanica is exclusively found in waters above 50 metres of depth, they are in theory well protected from bottom trawling, although occasional trawling is supposed to occur (Personal communication, NGO representative, April 2016).

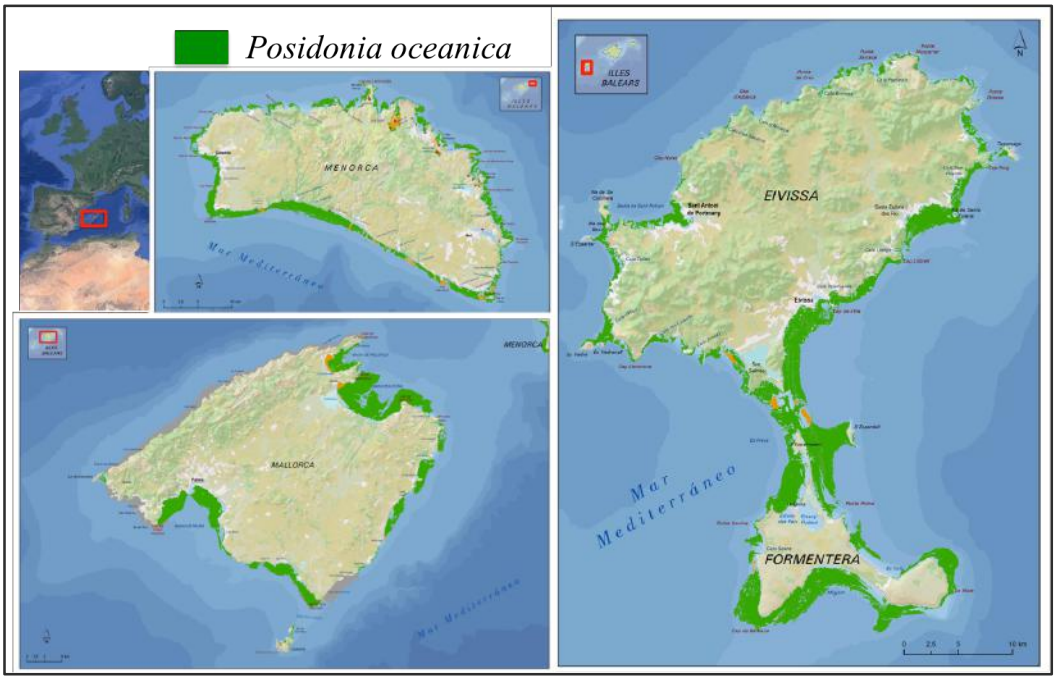

Figure 4 - Map of Balearic Islands and Posidonia oceanica meadows. Source: Ruiz, J.M., E. Guillén, A. Ramos Segura & M. Otero. 2015.

The European Union has engaged a new strategy to conserve and recover the ecological quality of the marine environment. In order to attain this objective, the Water Framework Directive (WFD) has established the basis of policies for the monitoring, protection and enhancement of the status of aquatic systems in the Member States. The main goal of the WFD is to achieve (or maintain at least) a "good water status" for all the European waters by 2015. To this end, this directive defines the concept of ecological status as the quality of the structure and functioning of ecosystems associated with homogenous water bodies (Gobert et al., 2009). Moreover, Posidonia oceanica meadows are identified as a priority habitat type for conservation under the Habitats Directive (Dir 92/43/CEE).

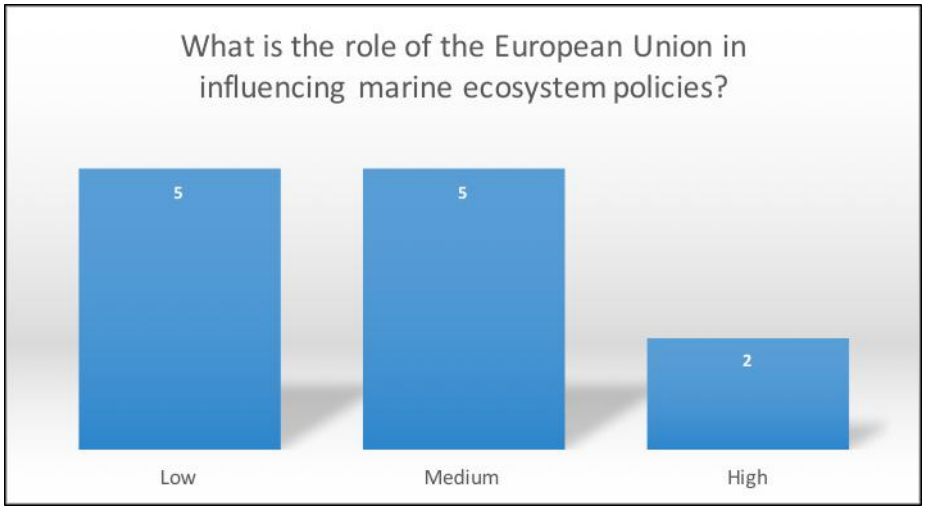

Figure 5 - Role of the EU in influencing policies

Jurisdictions involved and affected

The seagrass meadows around the island of Mallorca, which consist largely of the endemic Mediterranean species Posidonia oceanica, can be found down to a depth of approximately 45 meters. These underwater meadows "cross the border" between internal and external territorial waters. Therefore, both regional (internal waters) and national (external waters and nature reserve park) jurisdictions apply. Certain activities are, nevertheless, regulated and require the permission of local authorities to carry out these activities. For example, commercial fishing is an activity that although it mainly occurs in the external waters is still within the regional management of the Balearic Islands.

Property Rights arrangements

What can be owned?

Compared to terrestrial ecosystems and lands, the case of the marine environment is completely different as the sea and its resources cannot be owned by private entities. The Spanish Constitution declares that the coasts, beaches, territorial sea, internal waters and natural resources of the Exclusive Economic Zone (EEZ) and the continental shelf are of public domain. Therefore, everyone is entitled to make use and enjoy the coast.

Regarding fisheries in the EEZ – their use and exploitation are regulated by Spanish / Regional laws. In the context of the Baleares, bottom-trawling for instance is not allowed at depths equal or under 50 metres (or 1.5 miles from the coast), which means that most areas with Posidonia are protected from bottom trawling. However, there have been sporadic reports and instances that restriction of fishing and trawling are not always followed, attributed to a lack of control and enforcement. However, the estimation is that largely, trawling is does no longer occur in areas with Posidonia meadows.

Although Posidonia is not directly used, the use of other ecosystem services (i.e., fishing, recreational boating and anchoring on seagrass meadows) has direct effects on the health of the seagrass. In addition, land-based ecosystem services use (agriculture), tourism and recreation, indirectly impact seagrass meadows through the pollution (sewage) associated with these activities.

Who can own?

The state – hereby the regional autonomous government (internal waters) or the national state (external waters).

What can be done with it?

Since Posidonia oceanica is a species that is protected by Spanish law, the meadows itself cannot be used, and the plants cannot be harvested or otherwise commercially exploited.

User rights arrangements

The seagrass meadows are not directly used, except for recreational purposes, such as recreational fishing and scuba diving. It is prohibited by law to collect or damage the plants therefore activities such as boat anchoring on top of the seagrass meadows are also prohibited.

Understand the science

The Posidonia oceanica meadows cover approximately 650km^2 in the Balearic islands (represeting 50% of Spanish extension) (Alvaréz et al. 2015). The meadowns are an important ecosystem in the marine environment of the Balearic islands. In addition, Posidonia oceanica is an ecosystem engineer, widely used as a biological quality element in the context of the European Union Water Framework Directive (WFD) (Fraschetti et al., 2013). The seagrass meadows of Posidonia Oceanica provide numerous ecosystem services, such as erosion protection for beaches, attenuation of wave energy and beach replenishment. Seagrass meadows are an important habitat for many marine species and act as fish nurseries (Beck et al., 2001). Moreover, they are an important natural carbon sink (Fourqurean et al., 2012, Mazarrasa et al., 2015) that is in decline globally. Ecosystem services from Posidonia Oceanica, particularly maintaining water quality and transparency as well as the aforementioned beach replenishment and protection are vital for supporting the Balearic society and its main economic activity – tourism, which is heavily reliant on water quality and sandy beaches.

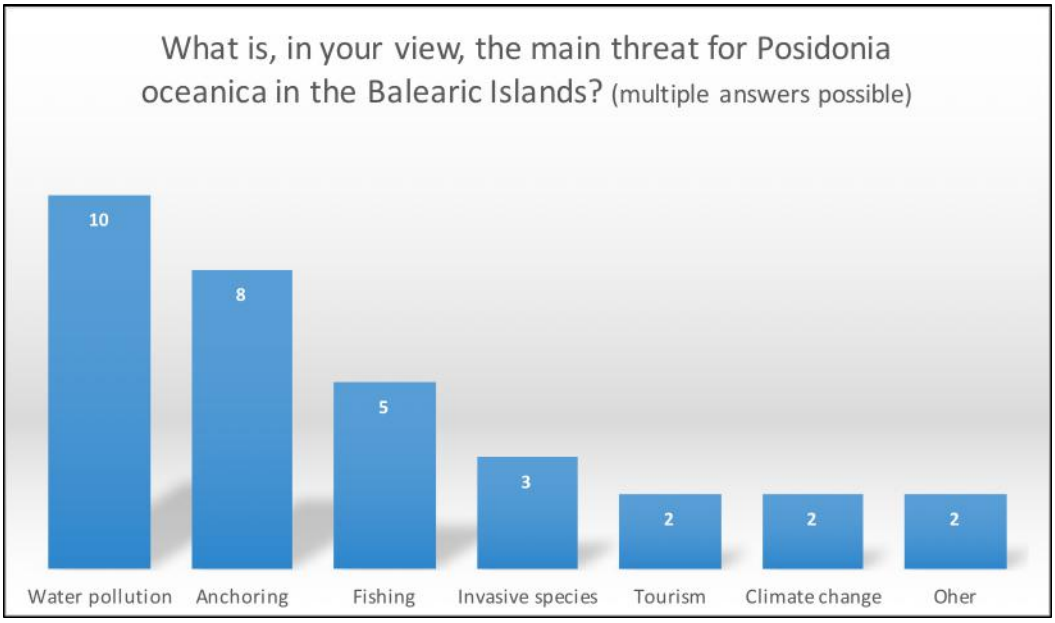

Figure 6 - Main threats to Posidonia oceanica in the Balearic Islands

According to interviewees, the main threat to Posidonia oceanica is perceived to be water pollution from land-based activities, such as agriculture and sewage flowing into the coastal waters. Anchoring of boats in Posidonia meadows is another direct threat, as anchors cast on the meadows destroy the plants and the root systems of the plants. With an increase in the number of recreational boats trafficking the waters of the Balearic Islands, anchoring close to beaches and in areas where the Posidonia oceanica meadows are located, the impact is expected to increase in the future.

Defined boundaries of the systems

The extent of the Posidonia seagrass meadows is estimated at 633 km^2 around the Balearic Islands. Nevertheless, it is hard to accurately estimate the real extent of the meadows due to the impossibility of remote sensing techniques to distinguish between different underwater habitats. However, the range of the ecological significance of these seagrass meadows is much larger than the area these currently cover. For instance, due to their significance as nursery grounds and their resulting importance for Mediterranean fish populations, the meadows are of fundamental importance for recreational and commercial fish species. Likewise, important drivers contributing to their decline are found outside the immediate ecosystem, for instance pollution from land sources or global climate change leading to temperature increases and ocean acidification which represents important stressors that reduce the resilience of the seagrass meadows. Other, more direct drivers for the decline in the seagrass meadows is the destruction through boat anchors (Balaguer et al., 2011).

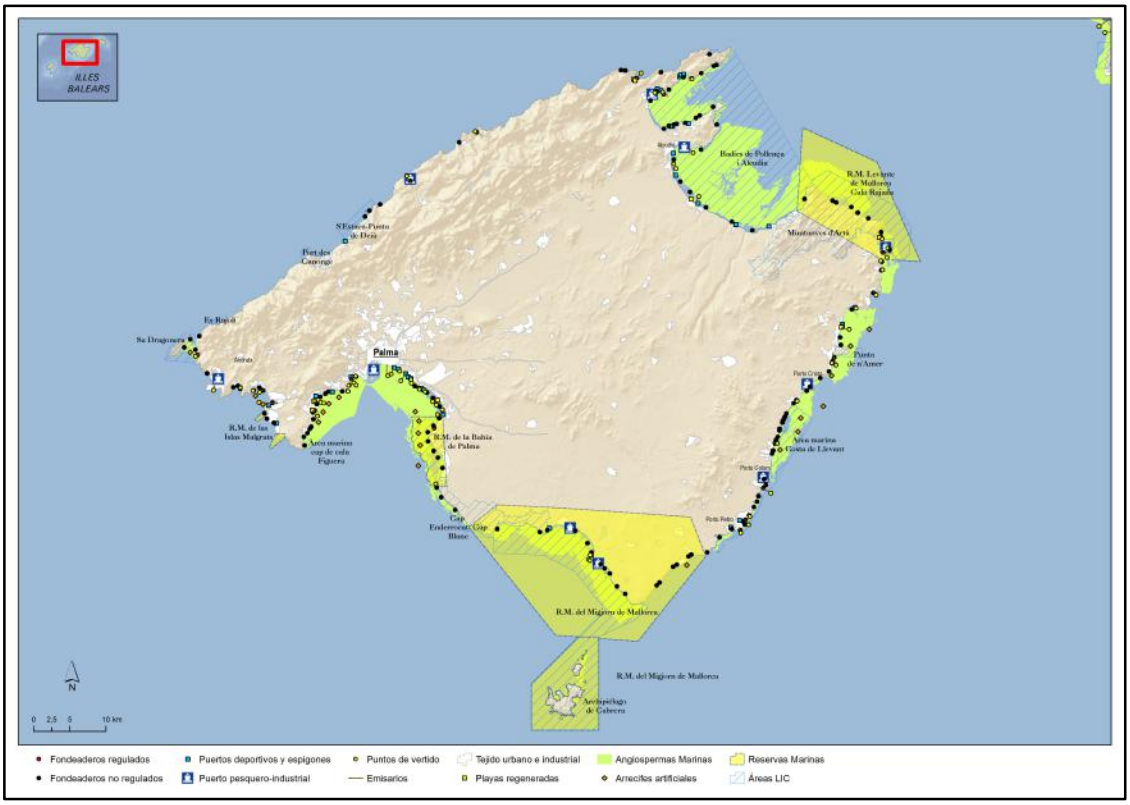

Figure 7 - Map of the Island of Mallorca indicating extend of Posidonia oceanica meadows (green), Marine protected areas and difference sources of pollution from sewage pipes (gray line), regulated (red dot) and unregulated anchoring (blue dot). Source: Ruiz, J.M., E. Guillén, A. Ramos Segura & M. Otero. 2015.

The blue dots along the coast of Mallorca in figure 8 indicate areas where boat anchoring (recreational boats, yachts) is largely unregulated and where anchoring poses a direct threat to the Posidonia meadows (see figure 9). In addition, particularly in the bay of Palma in the south and the bay of Pollenca in the north, sewage pipes that lead into the sea are a major contributor to water pollution.

Figure 8 - Boat anchor in Posidonia oceanica meadow in the waters of Formentera, Photo: Oceana

Temporal inertia and lags

One of the most important drivers of marine ecosystem change is climate change that leads to warmer water and ocean acidification, with negative consequences for Posidonia oceanica meadows (Jorda et al., 2012). In combination with the other drivers of degradation, such as pollution, anchoring and invasive species, the meadows are increasingly subsceptible to the negative impacts of climate change. Even if measures to protect Posidonia meadows from the other drivers were implemented, climate change would still pose a major threat.

Identifying stakeholders

In the context of the Balearic Posidonia oceanica meadows, the list of stakeholders is quite large, but the stakes undoublty vary from stakeholder to stakeholder. In Table 6 we present a list of institutions and stakeholder groups that are currently involved in marine ecosystem management in the Balearic Islands. The list is based on the interviews we conducted with policy and decision-makers and other experts in the spring of 2016 in Mallorca.

Table 6 - Institutions and stakeholder groups involved in marine ecosystem management in the Balearic Islands

Institutions and stakeholder groups (governmental, NGO's, others) that are currently involved in marine ecosystem management in the Balearic Islands							
Public administration		NGO's / Civil Society		Science		Private sector	
Local (Baleares)	Non-Local	Local	Non-Local	Local	Non- Local	Local	Non-Local
Fisheries authority	Central government (Demarcation of coasts)	The Balearic Ornithological Group (GOB)	Oceana	Balearic Islands Coastal Observing and Forecasting System (SOCIB)	Instituto Español de Oceanografía (IEO)	Palma Aquarium	International hotel groups
Authority for the Natural Environment	Ministry of defense (Water)	Consorcio de recuperacion de fauna de las Islas Baleares	Fundacion Ondine	Mediterranean Institute for Advanced Studies (IMEDEA)	Consejo Superior de Investigaciones Cientificas (CSIC)	Fishermen	
Environmental council	Foundation for biodiversity (Ministry of Environment)	Spanish Society for Ornithology (SEO) - marine birds	World Wildlife Fund (WWF)	Asociación Tursiops - Marine Research		Diving tourism	
Harbour authority	European Union	Fishermen association	Mission Blue	University of the Baleares		Nautical companies	
Marine reserves		Nautical association				Nautical clubs	
Authority for natural spaces and biodiversity						Federation of hotel businesses of Mallorca	
Council of the Island of Mallorca							
Tourism council							

At the local level – fishermen and people working in or depending on the tourism industry depend to a large degree on the health and extent of the seagrass meadows. Their role as nursery for numerous commercial and recreational fish species contributes significantly to the fishing industry. For the tourism industry, the seagrass meadows have a full range of benefits. Direct use include scuba diving and recreational fishing. In comparison to the overall number of tourists, it is probably only a minority of people who benefits directly from the seagrass meadows. The majority of tourists enjoys the indirect benefits Posidonia oceanica provides, chiefly beach replenishment, water quality and transparency and the prevention of beach erosion and wave attenuation.

Zooming out of the Balearic context – there are several regional and global benefits. Regionally, the nursery habitat for fish and other marine organisms has benefits beyond the Baleares. Globally, the capacity and ability of Posidonia oceanica to absorb and sequester carbon (blue carbon) is important in the mitigation of climate change.

Power relations among the stakeholders

The Baleares are heavily dependent on tourism, with an estimated 13 million tourist arrivals in 2016, the highest ever recorded number. Tourism is the regions biggest economic sector and the majority of the Balearic population is directly or indirectly dependent on tourism. It is therefore not surprising that the Balearic tourism industry is a particularly powerful stakeholder with regards to the political steering of the region.

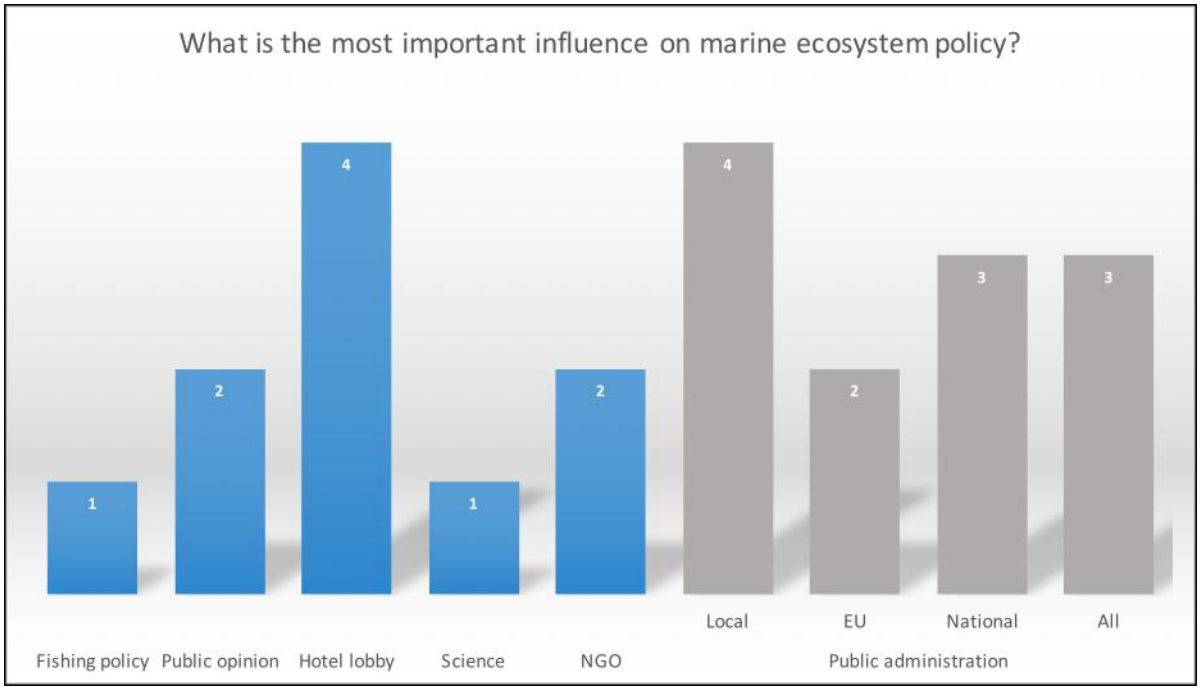

Figure 9 - Most important influence on marine ecosystem policy

According to our interviewees, the implications of the increase in tourism numbers and the important role and powerful voice of the sector, with vocal lobby groups such as as the hotel association and the recreational boating association, makes it difficult to implement progressive policies and environmental regulations that are interpreted as negative or limiting for the development of the Balearic tourism sector. For instance, althought it is legally prohibited to remove Posidonia oceanice from the sea and the dead plants from the beaches, local municipalities are removing Posidonia oceanica from the lcoal beaches in oder to maintain a clean and tourist friendly beach (see figure 10). The regional government grants special permits to the municipalities in order to do so and in the course of cleaning the beaches not violating the law (Royal Decree 139/2011).

Figure 10 - Removing Posidonia oceanica from the beach for the start of the tourism season. Picture taken by Torsten Krause, Bay of Palma, 2016

Production / distribution rules

The Posidonia oceanica meadows are not privately owned and are in both internal and external waters around the different Balearic islands. Most of the ecosystem services they provide are non-rival (water quality & transparency, beach replenishment and erosion prevention). These services are a pure public good. Other ecosystem services, such as the nursery function for many fish species are common property resources and they are to some extent rival, but also non-excludable.

Ideal typical mode of governance

According to our interviews and the review of the existing laws and regulations on marine environmental management and protection, the current legislation is deemed sufficient to protect Posidonia oceanica as a species and its marine environment. However, the general perception is that there is a lack of enforcement and control of existing regulations, such as the anchoring of boats. As one of our interviews has put it:

"The legal wrapping is there, but it is not being applied" (Personal communication, representative from local NGO, April 2016).

Awareness raising amongst the population of the Balearic islands and the tourists about the existence and benefits of Posidonia oceanica, both the living meadows and the dead plants on the beaches is considered important. Subsequently, enforcement of existing regulations and better coordination between the different public authorities is needed. Regulating anchoring of boats and enforcing the existing regulation alongside addressing the pollution from inadeuqate sweage treatment and waste water infrastructure is another component in the strategy to better manage and conserve the marine ecosystem and the Posidonia oceanica meadows around the Balearic isllands (see Figure 11).

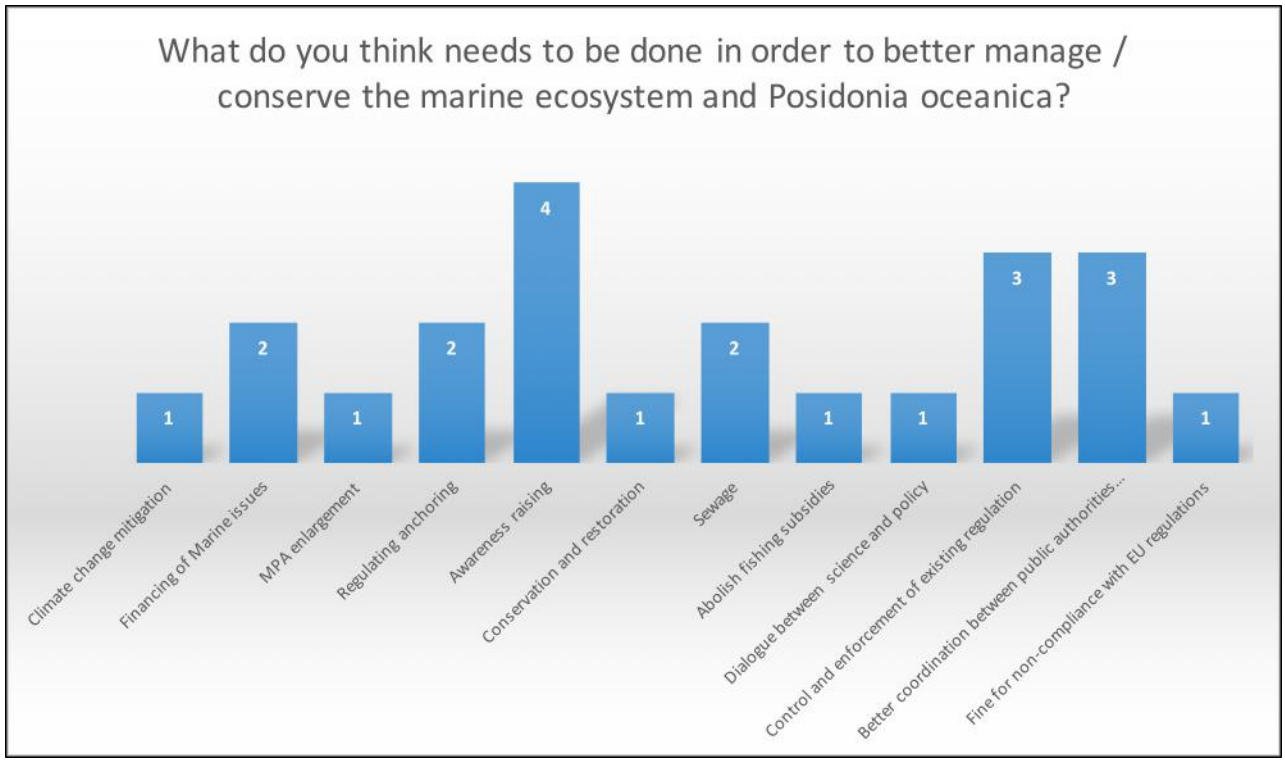

Figure 11 – Interviewees views on how to better manage and protect Posidonia oceanica around the Balearic Islands

Examples for ideal typical mode of governance

The management and conservation of Posidonia oceanica around the Balearic Islands reveals the conflicts bettwen different stakeholder groups around the governance of ecosystem services. The Island economy is dependent on income from tourism, which in turn is dependent on selling the beauty of the Islands beaches and water. However, the lack of awareness among the general population and the tourists create a perception that Posidonia oceanic is a nuisance when it is washed ashore and accumulates at the beaches. On the other hand, this ecological process is fundamental to maintain the

beaches of the Island and reduce beach erosion. This highlgihts the existing management issues that need to be addressed in the future. Apart from regulating the anchoring of boats and enforcing it, other measures that incentivize boat owners to anchor their boats in pre-defined places need to be sought in order to reduce the direct impact of boat anchors on the Posidonia oceanice meadows around the islands. This is particular important as the number of recreational vessels coming to the Islands is expected to increase further in the future. Current plans to install pre-paid fixed buoys are in palce, but they receive heave criticism from the nautical industry who is afraid that this will deter boaters to come to the Islands as the proposed plans undermine the freedom that boaters associate with the ability to anchor freely.

Another identified threat is water pollution, mainly through sewage pipes, but also through other more diffuse sources. Infrastructure investments to increase sweage treatment capacities and reduce the sewage and other waste water that flows into the sea can counteract the current pollution problem. The newly introduced Ecotax for tourists can provide the means to raise the necessary funds for infrastructure improvements and also for awareness raising campaings.

The interviews also reveales that the marine policies and mangement is compartementalized with a number of governmental institutions at the regional and national level. Our interviewees pointed out that the number of regulations is overwhelming and it is not clear who does what with regards to marine policy. Addressing the institutional overlap and tailoring marine conservation policy to the local context of the Baleares is key. One of our interviewees has highlighted that conservation policy for the marine environment is currently created with a land-based perspective and understanding at the national government level in Madrid. However, “there is no beach in Madrid” (Personal communication, regional government representative, April 2016), indicating that the national policies are far removed from the local realities and that the national government is, at the moment, not interested in finding a solutions for this problem.

6.2. The Scottish Multi-scalar Exemplar

The Scottish exemplar works on four scales, with different spatial and thematic focus: (1) building a science-policy-practice interface (ESCom), (2) a national assessment of ecosystem services and policy, (3) socio-cultural values of green space in peri-urban Edinburgh, (4) and local benefits of coastal wetlands realignment in the Inner Forth as well as social and cultural meanings of salt-marsh and mudflat restoration for local communities.

In our collaboration with the Scottish Exemplar, we focus on the Inner Forth area of Scotland. The Inner Forth is the roughly defined as the tidal range in the river Forth (Figure 16). The Inner Forth area is an old industrial and agricultural landscape influenced by the intertidal environments in the river Forth. A substantial part of the land currently used for industrial and agricultural purposes, has been claimed from the North Sea over the past 400 years. This led to the loss of over half of the tidal marsh and mudflat habitats, which are an internationally important breeding and wintering ground for wild fowl and waders. The industry, farmland, urban areas and tidal ecosystems are becoming increasingly vulnerable as climate change is posing a higher threats in terms of more frequent and severe coastal flooding and erosion, particularly during extreme weather events (Robins et al., 2016).

A managed realignment (MRA) refers a shift of the location of coastal defenses, making a allowing more or less land to be flooded at high tide. In the case of the Inner Forth, all MRA refers to the shift of coastal defenses landward, thus giving land back to the sea.

Figure 12 - Tidal habitat in the Inner Forth. Picture taken by Pontus Ambros, 2016

A managed realignment has several advantages as well as disadvantages that are briefly outlined here. Firstly, it can create an effective buffer against rising sea levels, particularly in areas that are low-lying (Zhu et al., 2010). Increasing the size of the intertidal zone has also proved to be effectively reducing coastal erosion, through natural sediment dynamics and absorption of wave power (Doody, 2013).

Second, MRA increases the area of habitat for a variety of estuarine species, as well as many species of wading birds that feed on these. In the Inner Forth, creating habitat for birds is an important argument for the restoration of tidal lands, as they receive large attention and the Royal Society for the Protection of Birds (RSPB) is one of the most active NGO's in the area. Yet, managed realignment has also direct social downside. Foremost the loss of good agricultural land that is farmed for generations is seen as critical (Ledoux et al., 2005).

The cost of implementing a single managed realignment scheme is high, posing a notable barrier to a landscape-scale uptake of nature-based solutions for flood management in the Inner Forth. Nevertheless, over a longer time period, managed relaginment can be cheaper than traditional hold-the-line solutions (Turner et al., 2007). However, there is an immediate trade-off with agricultural production as giving up agricultural land is not a preferred option by the owners of land (Ledoux et al., 2005). High-cost investments and compensation in the form of payments (either buying the land, or perpetual payment when land becomes flooded – for instance the multifunctional farming approach used in the Netherlands) are probably needed to support nature-based adaptations to rising sea levels and other climatic changes (i.e., more severe and unpredictable storms with associated flooding events).

A series of surveys and workshops have been completed in four towns around the Inner Forth (Airth, Alloa, Grangemouth and Kincardine) to create a shared community-driven vision for coastal management in the area. The methodological approach involves a combination of a choice experiment for coastal land use and management, deliberative mapping of the current landscape and potential future uses, and conceptual mapping of future drivers of change that residents are concerned about. The purpose of the survey and workshop activities is to understand the cultural and social values of the coastal marsh, and the preferences the residents hold for coastal management.

The preliminary surveys suggest that residents in the area have a low level of awareness about the current flood risk, expected changes in climate and sea level rise. There is also a low understanding for flood regulating benefits of intertidal habitats, such as tidal marshlands and mudflats. A high proportion of the workshop participants report their preferences for coastal land use and management to have changed after learning about the role that intertidal habitats play in coastal flood and erosion regulation. Overall, concerns over coastal wildlife and environmental health were generally seen as strong motivations for the restoration and enhancement of coastal marsh in the area. A notable group of residents are opposed to providing further access to the coastal nature areas due to the disturbance this might cause to tidal wildlife. Residents prefer future visions where conversion of land to marsh and improved access occur across the landscape rather than concentrated in a single area.

We asked what people who live around the Inner Forth think about the governance of the proposed land-use changes. Many workshop participants and citizens seem to hold local industries and the local and national government responsible for environmental degradation. Many respondents suggested that it is the industry's duty to pay for ecological improvements in the area of the Inner Forth. A preliminary analysis of the workshops that were held shows that the views on who should fund restoration changes in coastal land use. Overall, government and the private sector are often mentioned as being responsible,

but there is no consensus among the workshop participants on who should be responsible for addressing environmental degradation and vulnerability in the Inner Forth.

Yet, there are two things that most people seem to agree on:

1) A lot needs to be done (or should be done) in the coastal area, highlighting that people are concerned and want to see a shift away from the business as usual, and
2) Others (people and politicians) do not care enough, or are not able to contribute financially (at least that is the general excuse) - so there seems to be a negative social (community-level) barrier to instigate bottom-up changes and movements for restoration and realignment, i.e. a feeling of not being able to make a difference as a community.

Quite a large share of the lands that are identified as suitable for MRA is currently owned by private landowners who mostly use it for agriculture and to a minor extent also for nature conservation (Figure 16). We contacted 16 private landowners out of which ten agreed to be interviewed. Seven of the interviewees referred to themselves as farmers. Another portion of land was owned by a charitable trust, while another land holding is part of an estate belonging to the Earl of Mar. In addition to the nine agricultural land holdings we studies, one area is owned by an UK registered NGO that works with nature conservation. Both the charitable trust and the estate are managed through a land manager, leasing the land to local farmers. Most farmers had more than 50% of their income generated from their land and the charitable trust gains its income from leasing their land and buildings to tenants and agricultural contractors. All private landowners had a significant part of their income generated from the land, with exception from the NGO and the estate, (Ambros, 2016)(Land manager for Earl of Mar, Personal Communication, October 2016). Interviews with farmers showed that they were reluctant of giving up land, mainly due to two reasons, the first being the loss of income and secondly being the perception of cultural heritage. The cultural heritage refers to the feeling of being morally obliged to hold the reclaimed land and not retreating it to the sea (Ambros, 2016). This attitude is not shared by the estate, who's land manager and owner is positively minded to managed realignment on their land (Land manager for Earl of Mar, Personal communication, October 2016). Two councils own land along the Inner Forth and they are both positive to alternative flood protection of those specific sites (Ambros, 2016). The NGO who owns land along the river has already done a MRA on their land and is actively working to transform more land into MRA sites (Ambros, 2016).

Intersecting policies at EU, national and local level

Several different protective legislations are in place in the intertidal parts of the Inner Forth (Fig. 1). Large areas are simultaneously protected as Sites of Special Scientific Interest (SSSI) and Natura sites (Special Protection Areas (SPA) and Special Areas of Conservation (SAC)). The Birds Directive (2009/147/EC) is used to designate SPA while the Habitat Directive (92/43/EEC) are used to designate Special Areas of Conservation (SAC). Together they are commonly referred to as Natura sites, a common nature protection scheme for all of the European Union. Currently there are only SPA and SSSI sites within the study area. Separate from the natura sites, the Nature Conservation (Scotland) Act 2004, is effectively giving support to the SSSI's, a Scottish protection scheme for unique nature environments.

All of these protection schemes embodies the agreements from the RAMSAR convention of 1971 and all RAMSAR sites in Scotland is protected by at least one of these three conservation schemes (SNH, 2016).

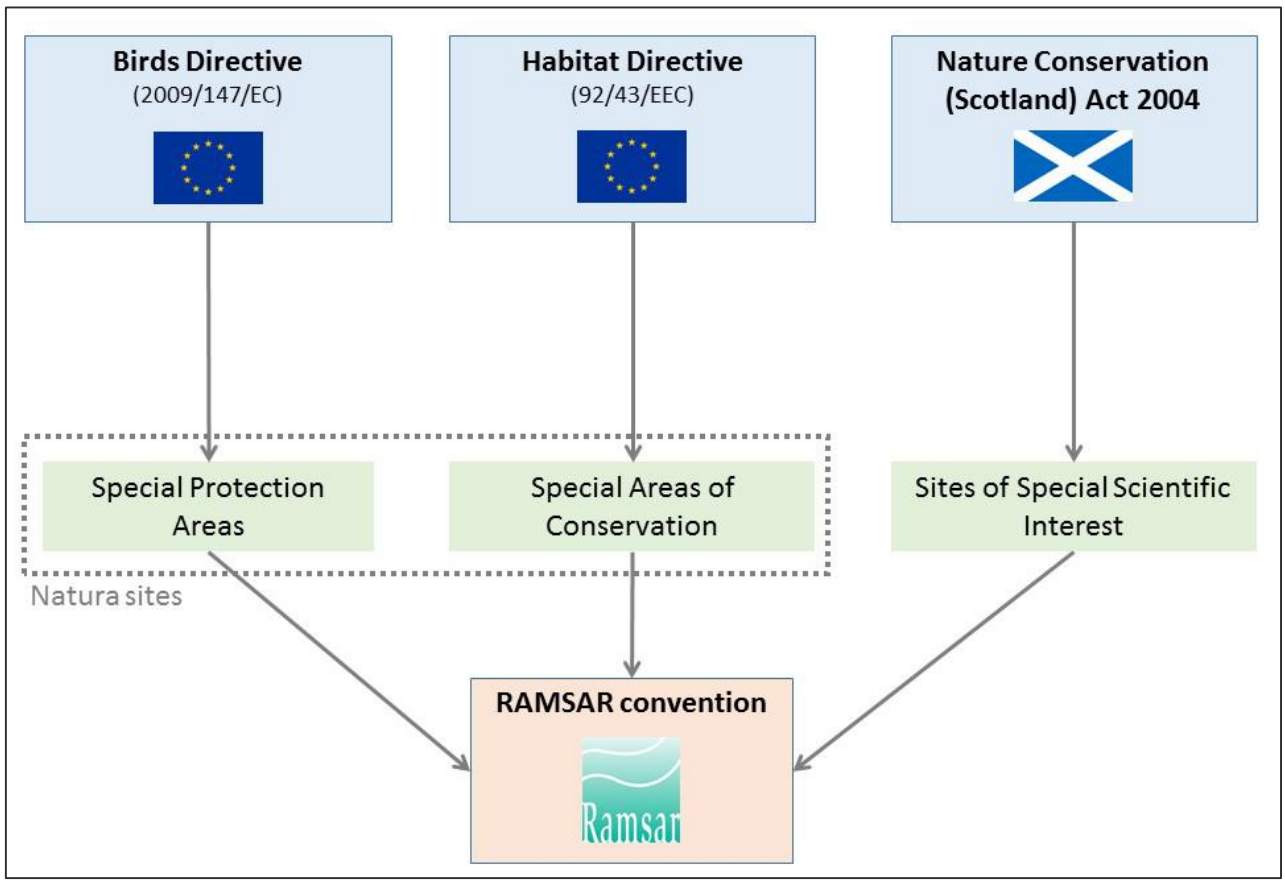

Figure 13 – Legislation in place for the protection of the intertidal habitats in the Inner Forth

If establishing new sites for MRA, both the Water Framework Directive (2000/60/EC), Flood Risk Management (Scotland) act (2009)[1] and the Climate Change (Scotland) act (2009)[2], would support such an initiative (Fig. 2) as these acts and directives can be used to actively support a managed realignment. However, the much older Coast Protection (UK) act (1949) might affectively work against a managed realignment, as it supports a static defence of the coast[3]. The Coast Protection (UK) act (1949) allows landowners to maintain the static sea defences, in order to protect their land from coastal floods and erosion. This contradictions in new versus old legislation can be used to both support MRA as well as keeping the current static defences. This duality giving the landowner the right to decide which type coastal defence to be used. Nonetheless, the councils have the right to step in and decide over the coastal defence using the Flood Risk Management (Scotland) act (2009) and actively take charge over a landowner's flood defence, either by forcing a certain defence strategy or making a forced buyout of the

[1] SEPA representative, personal communication, October 2016
[2] SNH representative, personal communication, October 2016
[3] SEPA representative, personal communication, October 2016

land needed to ensure the flood protection[4]. Although possible, there are no known cases in the Inner Forth where such a forced action has been taken. An important aspect highlighted by several stakeholders, is that the contradiction between the supporting and undermining legislation, is mostly theoretical. A MRA would require support from both the initiating organization, local authority (council), SEPA, SNH and landowner, before it can be implemented. Since all of these stakeholders then most likely agree of a coastal adaptation strategy, any contradictions would be irrelevant.

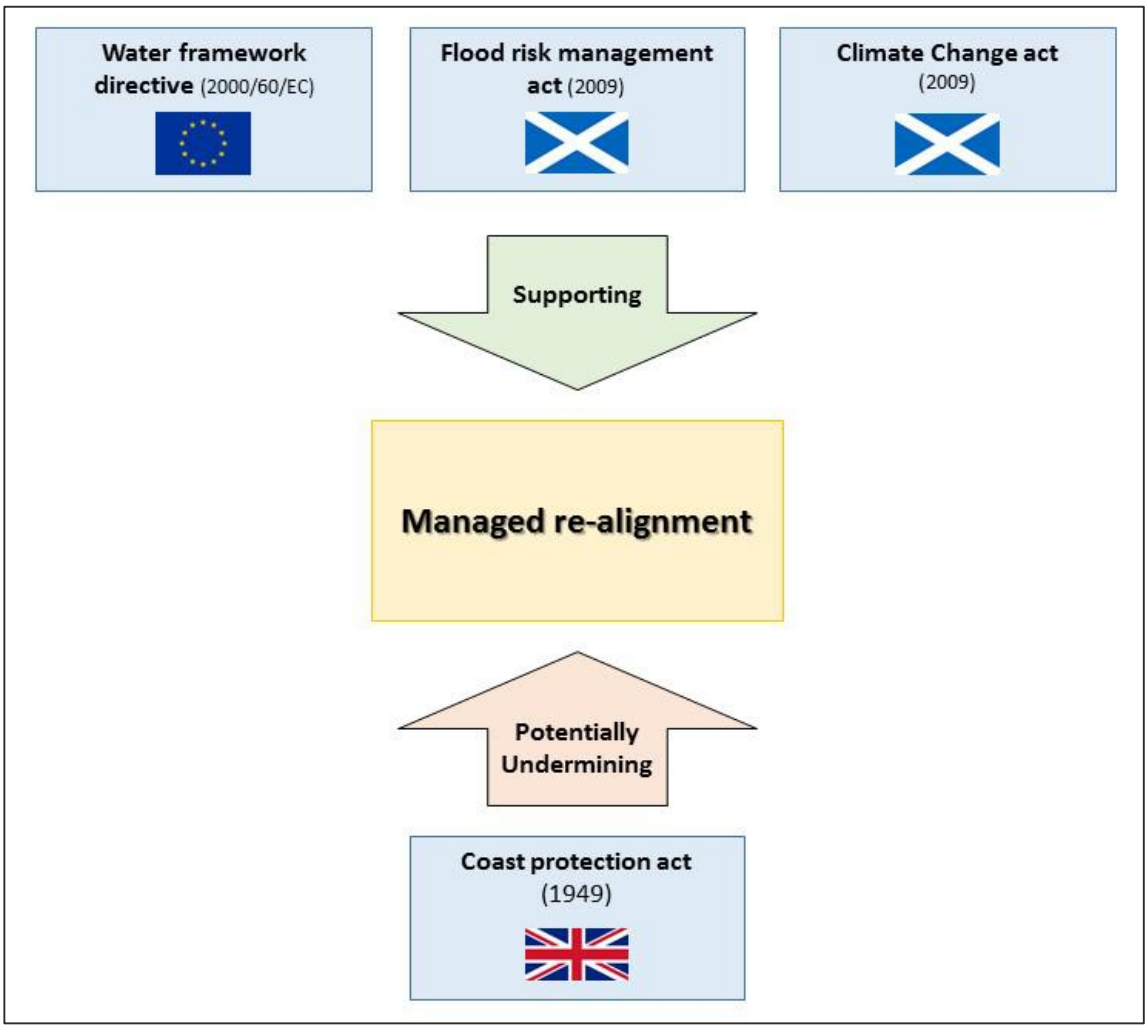

Figure 14 - Current legislation that can support and potentially also undermine managed realignment in Scotland

Jurisdictions involved and affected

There are two non-politically bound governmental organizations providing stakeholders with advice on nature conservation and flood protection (Figure 15). The first one is Scottish National Heritage (SNH),

[4] SEPA representative, personal communication, October 2016

who mainly works with maintaining and monitoring protected sites, such as SSSI, SPA and SAC, as well as advising the Scottish Government on legislation, etc. Beyond that, they also have an advisory role for local authorities (councils) and other interest groups working with nature conservation (SNH representative, personal communication, October 2016). The second governmental organization is the Scottish Environmental Protection Agency (SEPA), who works with flood protection, among other things. Similar to SNH, the Scottish Environmental Protection Agency also provides advice for local authorities (councils) and has an advisory role for the Scottish government. Based on the advice of the SNH and SEPA, the Scottish Government is issuing laws concerning environmental and flood protection (SEPA representative, personal communication, October 2016) (SNH representative, personal communication, October 2016). These laws are then implemented across Scotland, affecting landowners, councils, among others. Other legislation origin from European Union directives or the common law for the United Kingdom, , one such law is the Coast Protection (UK) act (1949). In some cases the local authorities are required to report back to the Scottish government.

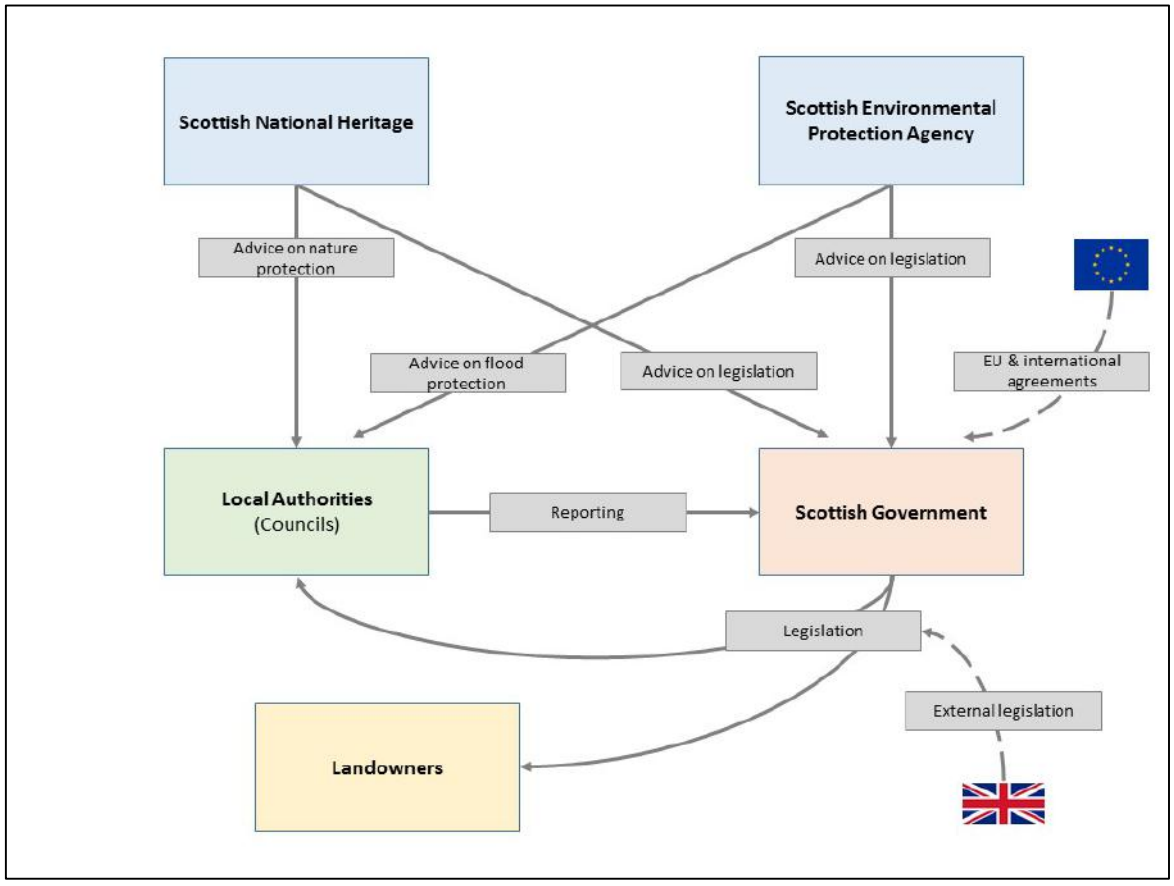

Figure 15 - Stakeholders affected by jurisdiction and being advised by SEPA and SNH.

Property Rights arrangements

The property rights of the Inner Forth are simplest explained by differentiating between the terrestrial shoreline (above the high water mark) and the intertidal zone (between the low and high water marks). A majority of the intertidal land is mudflats and saltmarshes owned by either the crown estate, private landowners or local authorities (councils). Although some of these sites are under private ownership, the development and management of them are highly restricted, due to their designation as SPA's and SSSI's.

The terrestrial land (above the high-water mark), lacks the same type of protection and therefore allows a greater degree to management and development. A majority of the terrestrial areas along the Inner Forth is under private ownership (see Figure 16). As the plans of a managed realignment is only being applicable in a rural context. All urbanized areas, such as villages, towns and industrial sites, were excluded from the study. The remaining sites are mostly agricultural land, nature reserves, and industrial waste sites. Most of these sites are privately owned by farmers, the Earl of Mar and a charitable organization that leases their land for agricultural activities. A few sites are owned by the local authorities (councils) and two minor sites are owned by non-governmental organizations, for nature conservation purposes.

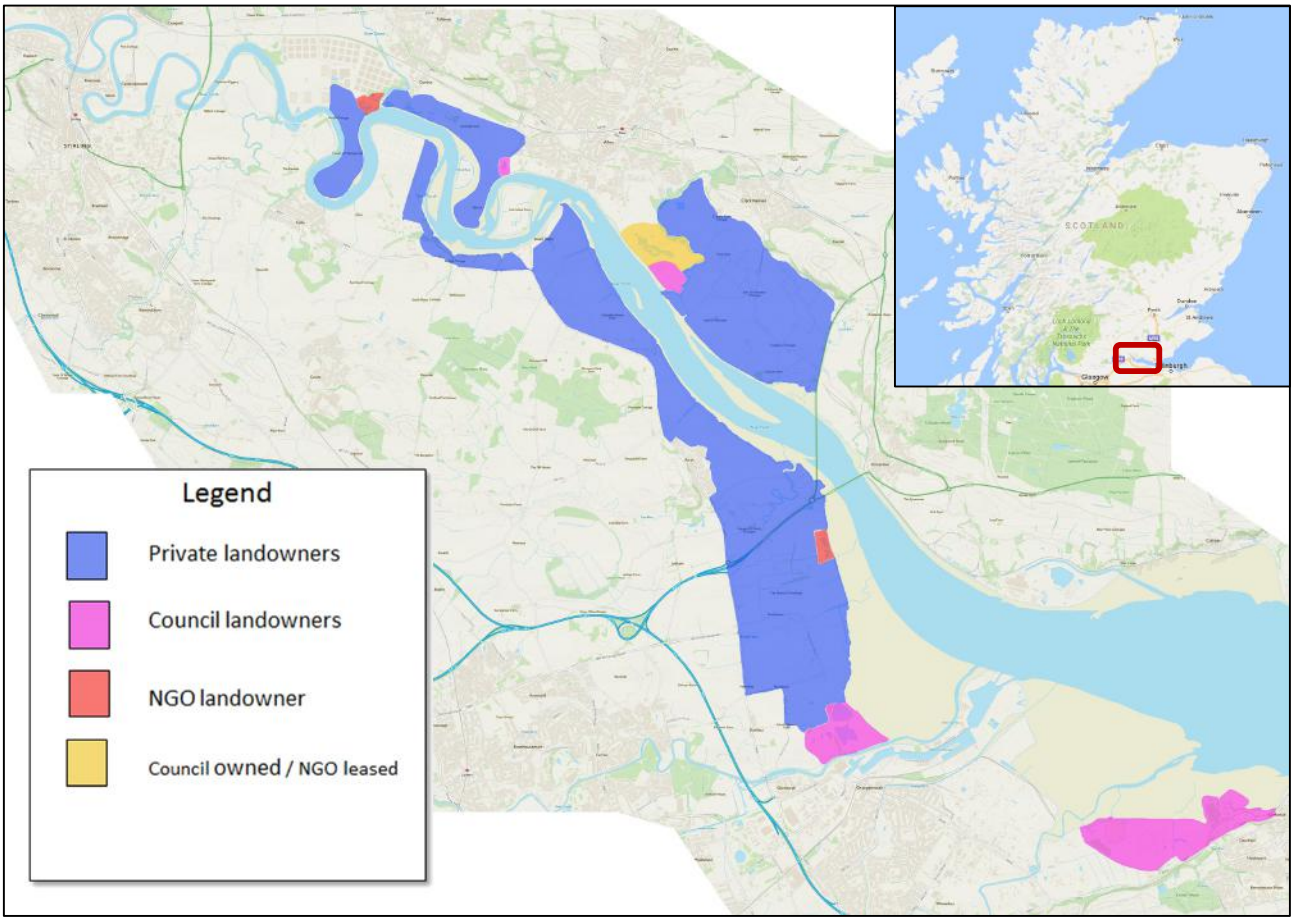

Figure 16 – Rough estimate of landownership in the study area of the Inner Forth. Urban areas and industrial sites are excluded from the study area. Map is only showing terrestrial landowners. Beige colour

indicates existing intertidal habitats. Contains OS data © Crown copyright and database right (Ordinance Survey data, 2016).

What can be owned?

There are no restrictions of landownership in the Inner Forth, as both terrestrial and intertidal sites can be owned by any landowner.

Who can own?

Any type legal entity can own both intertidal and terrestrial sites.

What can be done with it?

Terrestrial sites can be used and managed freely, as long as it full-fills the requirements of the current legislation.. The intertidal zone, that consists of mudflats and saltmarshes are under strict protection under the SPA and SSSI, while at the same time being designated RAMSAR sites. There are therefore strong restrictions in terms of management and development of the intertidal zone.

How can ownership be transferred?

Ownership of terrestrial land and intertidal land can be purchased, leased and sold, as normal. This also include land protected under by SSSI and land protected as Natura sites (SPA & SCA) (Sundseth and Creed, 2008). Councils can also do forced buyouts under the Flood Risk Management (Scotland) act (2009), if they see a certain land deemed essential for local flood protection. SNH can also under the Nature Conservation (Scotland) act (2004) force a buyout of land, if land is essential for nature conservation purposes. In both cases, the Scottish ministers are the judicial authority.

User rights arrangements

Land can either be owned or leased. There are several cases in the Inner Forth, where landowners lease the land to farmers, instead of utilizing it for their own purposes. Land lease contracts are usually active over a few years' period and often renewed when expired. The public has user rights to private owned land in form of freedom of roam. The freedom of roam is supported by the Land Reform (Scotland) act (2003) and makes Scotland unique in comparison with the other countries within the United Kingdom. As a result of the freedom of roam, some of the councils in the Inner Forth, cooperates with landowners to improve the access to privately owned land and thereby also reduce the damage caused to the agricultural production and wildlife in the area.

Understand the science

In the Inner Forth context, a MRA is mainly backed by the benefits of flood protection and nature conservation, both of which are largely understood by science SEPA is the governmental body in charge of advising on legislation for flood protection and local flood protection schemes (SEPA representative, personal communication, October 2016). They are also developing guidelines for coastal flood protection, such as the Natural Flood Management Handbook (SEPA, 2016). Their understanding of coastal flood management is well in line with what is being described by Barbier et al. (2011) and Zhu et al. (2010). SEPA is also contributing to the general scientific knowledge of coastal flooding and MRA, based on the Scottish context. It can therefore be assumed that this knowledge well is reflected in the Flood Risk Management (Scotland) act (2009) and the advises provided to the Scottish government and local authorities (councils). The scientific knowledge of the ecological values from the intertidal habitats are well understood, and so are the ecosystem services provided by these habitats (Barbier et al., 2011, Foster et al., 2013, Ma, 2005). Its threaten status is also somewhat understood by science (Kirwan et al., 2016, Hughes, 2004). Nonetheless, the overall trends show that these habitats are being threatened by human influence and climate change. One of the most obvious issues are the effects of coastal squeeze (when saltmarshes are decline due to being squeezed between the rising sea level and human infrastructure). This is a well-known phenomenon affecting intertidal habitats located along static structures such as seawalls and static infrastructure (Kirwan et al., 2016, Doody, 2004, Zhu et al., 2010). In the Inner Forth, a great majority of the coastline is boarded by static sea walls and is thereby posing a threat to the adjacent intertidal zone. SNH is well aware of these threats and is actively looking into the protection and improvements of the intertidal habitats. Its importance is also recognized by it being protection under SSSI and SPA. Interestingly, there are distinct difference between the new and old legislation, where newer legislation such as the Nature Conservation (Scotland) act (2004) and Flood Risk Management (Scotland) act (2009), works to strengthen these habitats, while older legislation, such as the Coast Protection (UK) act (1949), could on the other hand weaken these habitats, by supporting a static coastal defences strategy. To summarize, there is seemingly a good understanding of science in more recent legislation.

Defined boundaries of the systems

The study area of Inner Forth (Figure 16), is not yet defined by possible MRA sites. The boundaries are roughly connected to the geographical definition of the Inner Forth, which stretches from the end of the Forth estuary fjord (in height with Grangemouth) all the way up to the end of the tidal range of the river Forth (just before the town of Stirling). Within this area, low-lying land close to the river was included, with the exception of villages, urban settlements and industrial sites. However, managed realignment has been done in one area on the southern shore of the Inner Forth (Skinflats) by an national (UK) NGO – the Royal Society for the Protection of Birds (RSPB).

Temporal inertia and lags

As earlier mentioned, intertidal habitats are threatened by climate change and human development. Mainly due to coastal squeeze, but also because temperature changes and irregularities of weather patterns (Kirwan et al., 2016, Hughes, 2004). While some changes might be working in favour of the intertidal habitats, other changes will significantly reduce them (Hughes, 2004). However, the overall trends are pointing towards a decline (Kirwan et al., 2016, Ma, 2005). The terrestrial land is obviously threatened by sea level rise and coastal flooding due to storm surges and increased precipitation. A MRA is one beneficial adaptation strategy for those kinds of change (Zhu et al., 2010, Barbier et al., 2011). But when changing the land using a MRA, affectively alter the land from being terrestrial to intertidal. This bring changes to the hydrology of the site (Blackwell et al., 2004) and our interviews suggest that it might also impact adjacent land to some degree. We have come across some stakeholders, claiming that their land has been affected by the MRA already conducted in the Inner Forth. However, we are at the moment lacking scientific evidence to confirm that these changes can attributed to the MRA and we have not come across any studies researching the hydrological changes of adjacent land in connection of MRA. This could partly have to do with the fact that hydrological changes are hard to estimate as well as being site specific (SEPA representative, personal communication, October 2016).

Identifying stakeholders

In order to fully assess the stakeholders, three aspects need to be taken into account, (1) who are impacted, (2) who is influencing the development and (3) who might be benefitted and have an interest in the development and research (Durham et al., 2014). In the Inner Forth case, a broad group of stakeholders where identified based on those criteria, including; landowners, NGO's, individual citizens (as a group), businesses, local and governmental authorities. In the initial stakeholder mapping, the governance impact was estimated by looking at the three aspects mentioned earlier. Each stakeholder (or stakeholder group) was analysed on each of the three aspects and grade to have a low, low-medium, medium, medium-high or high impact (Figure 17).

Organisation	Type of org.	Role in the IF	Affected	Affected comment	Influence	Influence comment	Interest	Interest comment	Governance score
Example organization	NGO	NGO active in the IF region working with environment.	Low	Are not very affected by the development.	Medium	Has a medium influence in the development, due to strong lobbying.	High	Highly interested in the development and the reserach done in connection to it.	6

Figure 17 - example of the stakeholder analysis, showing level of affect, influence and interest. All summarized to a "Governance score".

Power relations among the stakeholders

To better understand the power relations between the stakeholders, a "Governance score" was estimated based on the level of impact, influence and interest. Their power in the respective categories was estimated using a five scale level system, where a low level generated 1 point, low/medium level generated 1.5 points, a medium level gave 2 points, a medium/high level gave 2.5 points and a high level generated 3 points (Figure 17). It should be noted that this method is not applicable for estimating the actual strength or impact of stakeholder. Although it can be used to better understand and visualize the stakeholder's relation to one another.

Highlighted by this analysis, it shows that many farmers are medium or highly impacted by the changes and development in the Inner Forth while still having little influence over the process. Councils and governmental organizations are at the same time highly affected but also highly influential (Figure 18). It should be noted that many farmer have a low interest in the development and landscape changes, which might also explains their low influence (Ambros, 2016). Our analysis show that there are some distortion in the relationship between impacted and influenced stakeholders. Where some, mainly farmers, are medium or highly impacted while only having a low or medium influence (Figure 18).

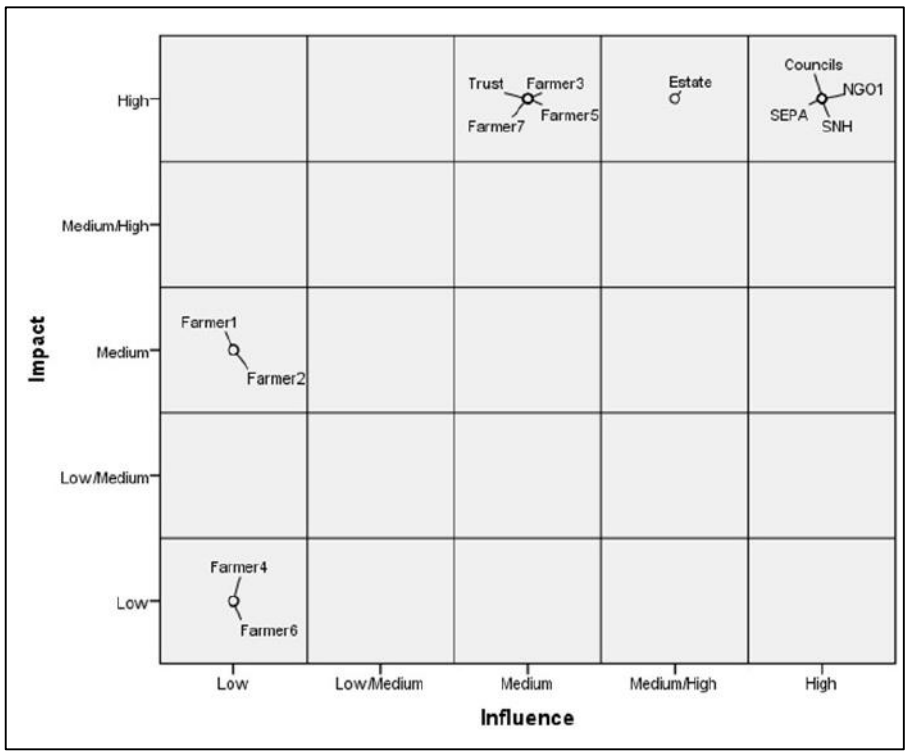

Figure 18 – Representation of the relation between impact and influence of stakeholders in the Inner Forth.

"We are trying not to get involved. Holding water back on the land as it has been suggested earlier is not a good way of doing it (speaking about the suggested creation of wetlands). Water needs to get out the land as soon as possible." (Personal communication, local farmer, March 2016)

Production / distribution rules

The identified sites for managed realignment in the Inner Forth are all located on currently used agricultural lands, and a great majority of them are privately owned, while a few sites are owned by local councils. Therefore the land and the provisioning services from it are both rival in use and laws are in place that regulate access to these services. Moreover, farmers are required by law to maintain the flood defenses themselves. With a proposal for managed realignment, some of the lands would be converted from a privately owned good to non-rival public goods, adding to the aesthetic value of the Inner Forth landscape and providing important habitat for biodiversity. In addition, the mitigation of flood risk through managed realignment has additional pure public good characted for the population of the Inner Forth.

Ideal typical mode of governance in the Inner Forth

Ideas for managed realignment as a soft engineering tool are not new and have been put forward for more than a decade in the UK (Ledoux et al., 2005, French, 2006). However, managed realignment is not yet an explicit policy goal and current legislation at the UK and the Scottish level can both support and hinder managed realignment schemes. Our study suggests that local councils are currently taking a sit-back and wait position on managed realignment. Despite different organisations such as the Inner Forth Landscape forum being in place, managed realignment does not feature highly in their work and neither do they reach the landowners whose lands are most affected by sea level rise and where managed realignment could potentially be carried out. This lack of communication is partly the consequence of a certain level of mistrust towards one of the leading NGO's who is a strong proponent of managed realignment. Asked about the willingness to be more involved in decision-making and discussions on flood management one of the interviewed farmers stated:

"I'm not sure to be honest, because the RSPB are fairly robust in their beliefs and as a land manager I would like to keep an arm's length from RSPB." (Personal communication, local farmer, March 2016).

Already in 2004 have Midgley and McGlashan (2004) that voluntary coastal partnerships are an important forum for deliberating about managed realignment schemes in coastal settings. Their study of the Forth Estuary Forum sheds light on the importance of local platforms and the involvement of the local communities in order to champion for managed realignment. Particularly public and civic bodies, as well as local communities need to come together in order to plan for future coastal management (Milligan et al., 2009), more so when the land in question is owned by a relatively small number of private landowners, as pointed out by one of our interviewees:

"There should be, at least the local farmers and landowners, but people who just stay in the country. They should all be involved in deciding in what's going to happened" (Personal communication, local farmer, March 2016)

Thus, under the current circumstances of low public stakeholder engagement and a lack of effective communication with landowners, a more deliberative communication and discussion forum is required in order to advance sustainable coastal management in the Inner Forth and to overcome the lack of trust that landowners exert towards conservation NGO's and any form of managed realignment.

Examples for ideal typical mode of governance

One of the proposed policies that might apply is the habitat compensation, meaning that a habitat is created or restored in order to compensate for the loss of this habitat type at another location. This can be done both after the habitat has been lost (retroactive) or prior to the loss (proactive, also known as "habitat banking"). The idea is that human development should have a "No Net Loss" of biodiversity (Moilanen et al., 2009). In England, this principal is commonly used when developing ports and urban infrastructure in conserved intertidal habitats, where the destruction of mudflats and saltmarshes are compensated by MRA at other sites (Brady and Boda, in press). But as pointed out by both Brady and Boda (in press) and Moilanen et al. (2009), it is complicated to sufficiently replace a habitat, due to several reasons. In the case of compensation of lost intertidal habitats, Brady and Boda (in press) points out several issues with the current system, showing that the compensation often does not fulfil the biodiversity requirements it been set up to maintain. Also the quality of newly established MRA sites, is likely to have lower outputs of ecosystem functions, thus not compensation fully for the loss of pristine sites (Mossman et al., 2012). In the case of Scotland, only one case of habitat compensation has been ratified and another is pending, none of these cases are compensating for intertidal habitats loss (SNH, personal communication, November 2016).

The number of landowners around the Inner Forth is relatively small. If managed realignment is to be adopted as a specific policy goal for flood risk management a direct payment or compensation scheme seems to be a feasible option for establishing managed realignment sites. Currently, adequate financial compensation to landowners for flood risk management is not in place. Moreover, it is unlikely that landowners will sell their land. More flexible approaches that are also favoured by landowners could take the form of easements or annual payments for flood risk management on their land (Beedell et al., 2012). Nevertheless, current legislation that requires landowners to maintain flood defenses and sea walls and holds them liable for damages related to poor sea wall maintenance is a major hurdle that needs to be addressed.

6.3. Summarizing the results from the two exemplars

The data we provided is based on written sources, publications and interviews with a range of stakeholders, policy-makers and experts in the Balaeric Islands and Scotland. In table 7 we present an overview of the findings for the two exemplars.

Table 7- Three selected exemplars and ideal types for the governance of ecosystem services, Adapted from Arnouts et al. 2012; Olsson et al. 2013.

	The Balearic Islands	**The Scottish Multi-scalar Exemplar (Focus on the Inner Forth area)**
Ecosystems studied	Marine - seagrass meadows	Tidal ecosystems, estuary, floodplains
Ecosystem service(s) in focus	Carbon sequestration, nutrient removal, fisheries	Flood prevention, habitat, cultural ecosystem services (use and non-use)
Scale (approximate size of area under investigation)	Regional (5,000 km^2), Posidonia oceanica meadows around the Balearic Islands approximately 633 km^2	Regional (300 km^2)
Property rights organization	Open access / public good	Private property (agricultural lands), 4 out of 12 potential sites are public lands (council lands – Falkirk, Stirling Fife and Clackmannanshire)
Existing regulations / laws for use of ES?	Partly –, fishery laws, Habitat Directive (EU), Regional and National habitat laws, national law protecting Posidonia oceanica as a threatened species	Partly – RAMSAR convention and Habitat Directive (EU), Legal responsibility for farmers to maintain flood defenses, no legal responsibility to adapt to rising sea levels
Are the property rights arrangements clear?	State and regional jurisdiction - marine based	Clear for land under agriculture, privately owned and leased; mudflats belong to Crown Estates, but some are leased to RSPB
Are the user rights arrangements clear?	Unclear – public good, anchoring is a particular use, regulations not sufficiently enforced	Clear for lands under agriculture, private good, council lands are public lands, but are in some cases leased to farmers for agriculture
Do we understand the science?	High for carbon sequestration ES and ecosystem threats; Medium for the other ES and ecosystem dynamics	High for flood risk management and storm surges; Medium for potential impacts of managed realignment on local hydrology

Are the boundaries of the systems defined / definable?	Yes – but land based agriculture and sewage disposal are key drivers of ecosystem decline from land based sources	Clear boundaries – defined by sea levels and flood maps
Are there temporal inertia and lags?	Yes – but not well understood	Yes
Can the stakeholders be defined?	Yes	Yes
Are power relations among the stakeholders clear?	Yes	Yes
Actors	Mainly regional and state government actors such as departments for Biodiversity, Climate Change, Fisheries, Tourism and Water; Ports Authority, Coasts department, Municipalities & other government agencies	Selected mix of actors - Landowners (farmers), Municipalities Government agency regulate and monitors protected areas; Charities (RSPB) manage some coastal areas;
Power	Government, strong hotel and tourism lobby	Pooled – landowners and government
Rules	Government coercion (seldom applied)	Restricted cooperation
Ideal typical governance types	Hierarchical governance / co-governance	Closed co-governance
Examples for ideal typical governance	Enforcement of regulation of anchroing for boats, penalties for non compliance, awareness campaigns, infrastructure improvements for sewage treatment	Land purchase, easements, payments for flood risk management

7. Summary and Conclusion

We presented a brief overview of the history of governance for ecosystem services. Based on the two selected exemplars, (1) the Balearic Islands and the co-beneficairy management of seagrass ecosystems and (2) coastal wetland realignment in the Inner Forth in Scotland, we developed a set of questions that serve as a guideline to study and unpack the different components that play a role for the governance of ecosystem services in the respective exemplar context. These questions are used to guide the creation of ideal types of governance modes for the selected ecosystem services. However, there are still a number of incoherencies and uncertainties with regards to the governance and the institutional context through which ecosystem services can and should be managed. This is not surprising as our understanding of the interlinkages of ecosystem functions, environemtnal changes and human actions evolves. Spatial scales vary greatly and temporal inertia and lags are often not sufficiently understood. And even if they are understand to a large degree, scientific evidence is frequently not sufficiently included in policies and management practices. This can be attributed to strong interest groups and other interests that play down scientific advise. In the case of the Balearic Islands, the hotel and tourism lobby is influential in shaping local environmental management that often supercedes available knowlede and advice. In addition, as part of a more inclusive form of governance for ecosystem services, platforms that give space to open deliberation and discussion regarding ecosystem management is key in the Inner Forth, where private landowners are most affected by sea level rise and managed realignment, but also the wider population who will be affected by coastal flooding in the Inner Forther area.

8. References

ABBOTT, K. W. & SNIDAL, D. 2009. The Governance Triangle - Regulatory Standards Institutions and the Shadow of the State. *The Politics of Global Regulation.* Princeton University Press.

AMBROS, P. 2016. Bridging to the common ground, adapting to climate change through sustainable estuarine land use: a study of the Inner Forth, Scotland. *Master Thesis Series in Environmental Studies and Sustainability Science*.

ARNOUTS, R., VAN DER ZOUWEN, M. & ARTS, B. 2012. Analysing governance modes and shifts - Governance arrangements in Dutch nature policy. *Forest Policy and Economics,* 16**,** 43-50.

BALAGUER, P., DIEDRICH, A., SARDÁ, R., FUSTER, M., CAÑELLAS, B. & TINTORÉ, J. 2011. Spatial analysis of recreational boating as a first key step for marine spatial planning in Mallorca (Balearic Islands, Spain). *Ocean & Coastal Management,* 54**,** 241-249.

BARBIER, E. B., HACKER, S. D., KENNEDY, C., KOCH, E. W., STIER, A. C. & SILLIMAN, B. R. 2011. The value of estuarine and coastal ecosystem services. *Ecological Monographs,* 81**,** 169-193.

BECK, M. W., HECK, K. L., ABLE, K. W., CHILDERS, D. L., EGGLESTON, D. B., GILLANDERS, B. M., HALPERN, B., HAYS, C. G., HOSHINO, K., MINELLO, T. J., ORTH, R. J., SHERIDAN, P. F. & WEINSTEIN, M. P. 2001. The Identification, Conservation, and Management of Estuarine and Marine Nurseries for Fish and Invertebrates: A better understanding of the habitats that serve as nurseries for marine species and the factors that create site-specific variability in nursery quality will improve conservation and management of these areas. *BioScience,* 51**,** 633-641.

BEEDELL, J., MORRIS, J. & HESS, T. 2012. Mobilising the Contribution of Rural Land Management to Flood Risk Management in Scotland. *Report to Scottish Government.* Peterborough.

BENNETT, A. B., MEEHAN, T. D., GRATTON, C. & ISAACS, R. 2014. Modeling pollinator community response to contrasting bioenergy scenarios. *PLoS ONE,* 9.

BENNETT, E. L. 2015. Another inconvenient truth: the failure of enforcement systems to save charismatic species. *Protecting the Wild.* Springer.

BERKES, F., COLDING, J., and C. FOLKE (eds).2003. Navigating Social-Ecological Systems – Building Resilience for Complexity and Change. Cambridge University Press. Cambridge.

BIERMANN, F., BETSILL, M. M., GUPTA, J., KANIE, N., LEBEL, L., LIVERMAN, D., SCHROEDER, H. & SIEBENHUNER, B. 2009. Earth System Governance: People, Places and the Planet. *Earth System Governance Project Report.* Bonn: Earth System Governance Project.

BLACKWELL, M. S. A., HOGAN, D. V. & MALTBY, E. 2004. The short-term impact of managed realignment on soil environmental variables and hydrology. *Estuarine, Coastal and Shelf Science,* 59**,** 687-701.

BLUMER, H. 1986. *Symbolic interactionism: Perspective and method*, Univ of California Press.

BRAAT, L. C. & DE GROOT, R. 2012. The ecosystem services agenda:bridging the worlds of natural science and economics, conservation and development, and public and private policy. *Ecosystem Services,* 1**,** 4-15.

BRADY, A. & BODA, C. in press. How do we know if managed realignment for coastal habitat compensation is successful? Insights from the implementation of the EU Birds and Habitats Directive in England. *Ocean & Coastal Management*.

BROWN, K. 2014. Global environmental change I: A social turn for resilience? *Progress in Human Geography,* 38**,** 107-117.

BULL, J. W., JOBSTVOGT, N., BÖHNKE-HENRICHS, A., MASCARENHAS, A., SITAS, N., BAULCOMB, C., LAMBINI, C. K., RAWLINS, M., BARAL, H., ZÄHRINGER, J., CARTER-SILK, E., BALZAN, M. V., KENTER, J. O., HÄYHÄ, T., PETZ, K. & KOSS, R. 2016. Strengths,

Weaknesses, Opportunities and Threats: A SWOT analysis of the ecosystem services framework. *Ecosystem Services,* 17**,** 99-111.

COAST PROTECTION ACT. 1949. 1949. Retrieved from: http://www.legislation.gov.uk/ukpga/1949/74/pdfs/ukpga_19490074_en.pdf [accessed November, 2016]

COMMISSION, E. 2013. CAP expenditure in the total EU expenditure (2007 constant prices). Brussels: DG Agriculture and Rural Develoment, Agricultural Policy Analysis and Perspectives Unit.

COSTANZA, R., D'ARGE, R., DE GROOT, R., FARBER, S., GRASSO, M., HANNON, B., LIMBURG, K., NAEEM, S., O'NEILL, R. V., PARUELO, J., RASKIN, R. G., SUTTON, P. & VAN DEN BELT, M. 1997. The value of the world's ecosystem services and natural capital. *Nature,* 387**,** 253-260.

COUNCIL OF EUROPE 2000. European Landscape Convention. Strasbourg: Council of Europe.

CRESSEY, D. 2013. Euope debates risk to bees. *Nature,* 496.

D'ADDA, G. 2011. Motivation crowding in environmental protection: Evidence from an artefactual field experiment. *Ecological Economics,* 70**,** 2083-2097.

DAILY, G. C., ALEXANDER, S., EHRLICH, P. R., GOULDER, L., LUBCHENCO, J., MATSON, P. A., MOONEY, H. A., POSTEL, S., SCHNEIDER, S. H., TILMAN, D. & WOODWELL, G. M. 1997. *Ecosystem Services: Benefits supplied to Human Societies by Natural Ecosystems* [Online]. Available: http://www.esa.org/sbi/issue2.htm [Accessed September 21, 2011].

DAW, T. & GRAY, T. 2005. Fisheries science and sustainability in international policy: a study of failure in the European Union's Common Fisheries Policy. *Marine Policy,* 29**,** 189-197.

DE GROOT, R. S., ALKEMADE, R., BRAAT, L., HEIN, L. & WILLEMEN, L. 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecological Complexity,* 7**,** 260-272.

DE GROOT, R. S., WILSON, M. A. & BOUMANS, R. M. J. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics,* 41**,** 16.

DIETZ, T., OSTROM, E. & P.C., S. 2003. The Struggle to Govern the Commons. *Science,* 302**,** 1907-1912.

DOBSON, A. 2003. *Citizenship and the Environment,* Oxford, Oxford University Press.

DOODY, J. P. 2004. 'Coastal Squeeze': An Historical Perspective. Opulus Press AB.

DOODY, J. P. 2013. Coastal squeeze and managed realignment in southeast England, does it tell us anything about the future? *Ocean & Coastal Management,* 79**,** 34-41.

DRYZEK, J. 1997. *The Politics of the Earth. Environmental Discourses.,* Oxford, Oxford University Press.

DURHAM, E., BAKER, H., SMITH, M., MOORE, E. & MORGAN, V. 2014. BiodivERsA Stakeholder Engagement Handbook. *BiddivERsA: Paris, France*.

ECKERSLY, R. 2004. *The Green State. Rethinking Democracy and Sovereignty.,* Cambridge MA, MIT Press.

EEA. 2016. Structure of CICES. European Environmental Agency. Retrieved from: http://cices.eu/cices-structure/ [accessed November 2016]

EHRLICH, P. R. & MOONEY, H. A. 1983. Extinction, Substitution, and Ecosystem Services. *Bioscience,* 33**,** 248-254.

EIKENBERRY, A. M. & KLUVER, J. D. 2004. The Marketization of the Nonprofit Sector. Civil Society at Risk? *Public Administration Review* 64**,** 132-140.

EUROPEAN COMMISSION 2011. Our life insurance, our natural capital: an EU biodiversity strategy to 2020. Brussels: COM(2011) 244.

EUROPEAN PARLIAMENT AND COUCIL 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. Brussels: Official Journal L 327, 22/12/2000 P. 0001-0073.

FARLEY, J. 2012. Ecosystem services: The economics debate. *Ecosystem Services,* 1**,** 40-49.

FISCHER, J., GARDNER, T. A., BENNETT, E. M., BALVANERA, P., BIGGS, R., CARPENTER, S., DAW, T., FOLKE, C., HILL, R., HUGHES, T. P., LUTHE, T., MAASS, M., MEACHAM, M., NORSTRÖM, A. V., PETERSON, G., QUEIROZ, C., SEPPELT, R., SPIERENBURG, M. & TENHUNEN, J. 2015. Advancing sustainability through mainstreaming a social-ecological systems perspective. *Current Opinion in Environmental Sustainability,* 14**,** 144-149.

FLOOD RISK MANAGEMENT (SCOTLAND) ACT 2009. 2009. Retrieved from: http://www.legislation.gov.uk/asp/2009/6/pdfs/asp_20090006_en.pdf. [Accessed November 2016]

FOSTER, N. M., HUDSON, M. D., BRAY, S. & NICHOLLS, R. J. 2013. Intertidal mudflat and saltmarsh conservation and sustainable use in the UK: A review. *Journal of environmental management,* 126**,** 96-104.

FOURQUREAN, J. W., DUARTE, C. M., KENNEDY, H., MARBA, N., HOLMER, M., MATEO, M. A., APOSTOLAKI, E. T., KENDRICK, G. A., KRAUSE-JENSEN, D., MCGLATHERY, K. J. & SERRANO, O. 2012. Seagrass ecosystems as a globally significant carbon stock. *Nature Geosci,* 5**,** 505-509.

FRASCHETTI, S., GUARNIERI, G., BEVILACQUA, S., TERLIZZI, A. & BOERO, F. 2013. Protection Enhances Community and Habitat Stability: Evidence from a Mediterranean Marine Protected Area. *PLoS ONE,* 8**,** e81838.

FRENCH, P. W. 2006. Managed realignment – The developing story of a comparatively new approach to soft engineering. *Estuarine, Coastal and Shelf Science,* 67**,** 409-423.

FREUND, J. 1967. The Sociology of Max Weber. Pantheon.

GARDNER, T. A., VON HASE, A., BROWNLIE, S., EKSTROM, J. M. M., PILGRIM, J. D., SAVY, C. E., STEPHENS, R. T. T., TREWEEK, J. O., USSHER, G. T., WARD, G. & TEN KATE, K. 2013. Biodiversity Offsets and the Challenge of Achieving No Net Loss. *Conservation Biology,* 27**,** 1254-1264.

GIDDENS, A. 2009. *The Politics of Climate Change,* Cambridge UK, Polity Press.

GOBERT, S., SARTORETTO, S., RICO-RAIMONDINO, V., ANDRAL, B., CHERY, A., LEJEUNE, P. & BOISSERY, P. 2009. Assessment of the ecological status of Mediterranean French coastal waters as required by the Water Framework Directive using the Posidonia oceanica Rapid Easy Index: PREI. *Marine Pollution Bulletin,* 58**,** 1727-1733.

GOMEZ-BAGGETHUN, E. & DE GROOT, R. 2010. Natural capital and ecosystem services: The ecological foundation of human society. *Issues in Environmental Science and Technology.*

GÓMEZ-BAGGETHUN, E. & RUIZ-PÉREZ, M. 2011. Economic valuation and the commodification of ecosystem services. *Progress in Physical Geography,* 35**,** 613-628.

GOODIN, R. E. 1992. *Green Political Theory.,* Cambridge, Polity Press.

GREIBER, T. AND S. SCHIELE. 2011. Governance of Ecosystem Services. IUCN, Gland, Switzerland.

GUNNINGHAM, N., KAGAN, R. A. & THORNTON, D. 2003. *Shades of green: business, regulation, and environment,* Palo Alto, Ca., Stanford University Press.

HAUCK, J., SCHWEPPE-KRAFT, B., ALBERT, C., GORG, C., JAX, K., JENSEN, R., FURST, C., MAES, J., RING, I., HONIGOVA, I., BURKHARD, B., MEHRING, M., TIEFENBACH, M., GRUNEWALD, K., SCHWARZER, M., MEURER, J., SOMMERHAUSER, M., PRIESS, J. A., SCHMIDT, J. & GRET-REGAMEY, A. 2013. The Promise of the Ecosystem Services Concept for Planning and Decision-Making. *Gaia-Ecological Perspectives for Science and Society,* 22**,** 232-236.

HOWLETT, M. 2009. Governance Modes, Policy Regimes and Operational Plans: A Multi-Level Nested Model of Policy Instrument Choice and Policy Design. *Policy Sciences,* 42**,** 73-89.

HUGHES, R. G. 2004. Climate change and loss of saltmarshes: consequences for birds. *Ibis,* 146**,** 21.

IEEP 2011. Cohesion Policy and Sustainable Development: Final Synthesis Report. London: Institute for European Environmental Policy (IEEP).

JAX, K., BARTON, D. N., CHAN, K. M. A., DE GROOT, R., DOYLE, U., ESER, U., GÖRG, C., GÓMEZ-BAGGETHUN, E., GRIEWALD, Y., HABER, W., HAINES-YOUNG, R., HEINK, U., JAHN, T., JOOSTEN, H., KERSCHBAUMER, L., KORN, H., LUCK, G. W., MATZDORF, B., MURACA, B., NESSHÖVE, C., NORTON, B., OTT, K., POTSCHIN, M., RAUSCHMAYER, F., VON HAAREN, C. & WICHMANN, S. 2013. Ecosystem services and ethics. *Ecological Economics,* 93**,** 260-268.

JESSOP, B. 2002. Liberalism, Neo-Liberalism and Urban Governance: A State Theoretical Perspective. *Antipode,* 34**,** 452-472.

JORDA, G., MARBA, N. & DUARTE, C. M. 2012. Mediterranean seagrass vulnerable to regional climate warming. *Nature Clim. Change,* 2**,** 821-824.

KHALILIAN, S., FROESE, R., PROELSS, A. & REQUATE, T. 2010. Designed for failure: A critique of the Common Fisheries Policy of the European Union. *Marine Policy,* 34**,** 1178-1182.

KIRWAN, M. L., TEMMERMAN, S., SKEEHAN, E. E., GUNTENSPERGEN, G. R. & FAGHERAZZI, S. 2016. Overestimation of marsh vulnerability to sea level rise. *Nature Climate Change,* 6**,** 253.

KOOIMAN, J. (ed.) 1993. *Modern Governance: New Government-Society Interactions,* London: Sage.

KOOIMAN, J. 2003. *Governing as Governance*, Sage.

KOSOY, N. & CORBERA, E. 2010. Payments for ecosystem services as commodity fetishism☆. *Ecological Economics,* 69**,** 1228-1236.

KUMAR, P. 2010. *The Economics of Ecosystems and Biodiversity: ecological and economic foundations,* London and Washington DC, UNEP and Earthscan.

KURANI, S. 2015. Forget about carbon – let's go on holiday. Using tourist values to conserve seagrass meadows. Lund University Master Thesis in Environmental Studies and Sustainability Science.

LAND REFORM (SCOTLAND) ACT 2003. 2003. Retrieved from: http://www.legislation.gov.uk/asp/2003/2/pdfs/asp_20030002_en.pdf. [Accessed November 2016]

LAUTENBACH, S., LEE, H., MUPEPELE, A.-C., DORMANN, C. F. & VOLK, M. 2013. MS 2.3 - Preliminary Report on Knowldge Gaps and Demand for Instruments. *OPERAs Milestone.*

LEDOUX, L., CORNELL, S., O'RIORDAN, T., HARVEY, R. & BANYARD, L. 2005. Towards sustainable flood and coastal management: identifying drivers of, and obstacles to, managed realignment. *Land Use Policy,* 22**,** 129-144.

LOWNDES, V. & SKELCHER, C. 1998. The Dynamics of Multi-Organizational Partnerships: An Analysis of Changing Modes of Governance. *Public Administration,* 76**,** 313-333.

LUCK, G. W., CHAN, K. M. A., ESER, U., GÓMEZ-BAGGETHUN, E., MATZDORF, B., NORTON, B. & POTSCHIN, M. B. 2012. Ethical Considerations in On-Ground Applications of the Ecosystem Services Concept. *Bioscience,* 62**,** 1020-1029.

LUNDIN, O., SMITH, H. G., RUNDLOF, M. & BOMMARCO, R. 2013. When ecosystem services interact: crop pollination benefits depend on the level of pest control. *Proceedings of the Royal Society B-Biological Sciences,* 280.

LUNDY, L. & WADE, R. 2011. Integrating sciences to sustain urban ecosystem services. *Progress in Physical Geography,* 35**,** 653-669.

MA, M. E. A. 2005. Ecosystems and human well-being: current state and trends. Island Press, Washington, DC.
MANLEY, R., BOOTS, M. & WILFERT, L. 2015. Emerging viral disease risk to pollinating insects: Ecological, evolutionary and anthropogenic factors. *Journal of Applied Ecology,* 52**,** 331-340.
MARTINEZ-ALIER, J., MUNDA, G. & O'NEILL, J. 1998. Weak comparability of values as a foundation for ecological economics. *Ecological Economics,* 26**,** 277-286.
MAZARRASA, I., MARBÀ, N., LOVELOCK, C. E., SERRANO, O., LAVERY, P. S., FOURQUREAN, J. W., KENNEDY, H., MATEO, M. A., KRAUSE-JENSEN, D., STEVEN, A. D. L. & DUARTE, C. M. 2015. Seagrass meadows as a globally significant carbonate reservoir. *Biogeosciences,* 12**,** 4993-5003.
MCAFEE, K. 2012. The Contradictory Logic of Global Ecosystem Services Markets. *Development and Change,* 43**,** 105-131.
MEA 2005. Ecosystems and Human Well-Being: Biodiversity Synthesis. *Millenium Ecosystem Assessment.* Washington, DC: World Resources Institute.
MELATHOPOULOS, A. P. & STONER, A. M. 2015. Critique and transformation: On the hypothetical nature of ecosystem service value and its neo-Marxist, liberal and pragmatist criticisms. *Ecological Economics,* 117**,** 173-181.
MIDGLEY, S. & MCGLASHAN, D. J. 2004. Planning and management of a proposed managed realignment project: Bothkennar, Forth Estuary, Scotland. *Marine Policy,* 28**,** 429-435.
MILLIGAN, J., O'RIORDAN, T., NICHOLSON-COLE, S. A. & WATKINSON, A. R. 2009. Nature conservation for future sustainable shorelines: Lessons from seeking to involve the public. *Land Use Policy,* 26**,** 203-213.
MOILANEN, A., VAN TEEFFELEN, A. J., BEN-HAIM, Y. & FERRIER, S. 2009. How much compensation is enough? A framework for incorporating uncertainty and time discounting when calculating offset ratios for impacted habitat. *Restoration Ecology,* 17**,** 470-478.
MOSSMAN, H. L., DAVY, A. J. & GRANT, A. 2012. Does managed coastal realignment create saltmarshes with 'equivalent biological characteristics' to natural reference sites? *Journal of Applied Ecology,* 49**,** 1446-1456.
MURADIAN, R. & RIVAL, L. 2012. Between markets and hierarchies: The challenge of governing ecosystem services. *Ecosystem Services,* 1**,** 93-100.
NANZ, P. & STEFFEK, J. 2005. Assessing the Democratic Quality of Deliberation in International Governance: Criteria and Research Strategies. *Acta Politica,* 40**,** 368-383.
NATURE CONSERVATION (SCOTLAND) ACT 2004. 2004. Retrieved from: http://www.legislation.gov.uk/asp/2004/6/pdfs/asp_20040006_en.pdf. [Accessed November 2016]
NORGAARD, R. B. 2010. Ecosystem services: From eye-opening metaphor to complexity blinder. *Ecological Economics,* 69**,** 1219-1227.
O'NEILL, J., HOLLAND, A. & LIGHT, A. 2008. *Environmental values,* New York, Routledge.
ODLING-SMEE, L. 2005. Conservation: Dollars and sense. *Nature,* 437**,** 614-616.
OLSSON, L., JERNECK, A., THOREN, H., PERSSON, J. & O'BYRNE, D. 2015. Why resilience is unappealing to social science: Theoretical and empirical investigations of the scientific use of resilience. *Science Advances,* 1**,** e1400217.
OSTROM, E. 2009. A General Framework for Analyzing Sustainability of Social-Ecological Systems. *Science,* 325**,** 419-422.
PAGIOLA, S., ARCENAS, A. & PLATAIS, G. 2004. Can Payments for Environmental Services Help Reduce Poverty? An Exploration of the Issues and the Evidence to Date from Latin America. *World Development,* 33**,** 237-253.
PLIENINGER, T., SCHLEYER, C., SCHAICH, H., OHNESORGE, B., GERDES, H., HERNÁNDEZ-MORCILLO, M. & BIELING, C. 2012. Mainstreaming ecosystem services through reformed European agricultural policies. *Conservation Letters,* 2012**,** 281-288.

POTSCHIN, M. B. & HAINES-YOUNG, R. H. 2011. Ecosystem services: Exploring a geographical perspective. *Progress in Physical Geography,* 35**,** 575-594.

PRIMMER, E., JOKINEN, P., BLICHARSKA, M., BARTON, D. N., BUGTER, R. & POTSCHIN, M. 2015. Governance of Ecosystem Services: A framework for empirical analysis. *Ecosystem Services,* 16**,** 158-166.

RHODES, R. A. W. 1997. *Understanding Governance: Policy Networks, Governance, Reflexivity, and Accountability,* Buckingham, Open University Press.

RIGG, J. D. 2006. Forests, marketization, livelihoods and the poor in the Lao PDR. *Land Degradation and Development,* 17**,** 123-133.

ROBERTSON, M. M. 2004. The neoliberalization of ecosystem services: wetland mitigation banking and problems in environmental governance. *Geoforum,* 35**,** 361-373.

ROBINS, P. E., SKOV, M. W., LEWIS, M. J., GIMÉNEZ, L., DAVIES, A. G., MALHAM, S. K., NEILL, S. P., MCDONALD, J. E., WHITTON, T. A., JACKSON, S. E. & JAGO, C. F. 2016. Impact of climate change on UK estuaries: A review of past trends and potential projections. *Estuarine, Coastal and Shelf Science,* 169**,** 119-135.

RODE, J., GÓMEZ-BAGGETHUN, E. & KRAUSE, T. 2015. Motivation crowding by economic incentives in conservation policy: A review of the empirical evidence. *Ecological Economics,* 117**,** 270-282.

RUIZ, J.M., E. GUILLÉN, A. RAMOS SEGURA & M. OTERO. 2015. Atlas de las praderas marinas de España. IEO/IEL/UICN, Murcia-Alicante-Málaga, 681 pp.

OLSSON, L., KRAUSE, T., HIRSCHI, C. AND M. KETTUNEN. 2013. Set of generic questions sent to selected exemplars regarding salient characteristics of ES/NC and stakeholders. OPERAS Milestone 3.6.

OS DATA. 2016. OS Maps. Retrieved from: https://www.ordnancesurvey.co.uk/ [accessed November 2016]

SACHS, J. D. 2005. *The End of Poverty, how we can make it happen in our lifetime,* London, Penguin.

SCHRÖTER, M., VAN DER ZANDEN, E. H., VAN OUDENHOVEN, A. P. E., REMME, R. P., SERNA-CHAVEZ, H. M., DE GROOT, R. S. & OPDAM, P. 2014. Ecosystem services as a contested concept: a synthesis of critique and counter-arguments. *Conservation Letters***,** n/a-n/a.

SCHULP, C. J. E., LAUTENBACH, S. & VERBURG, P. H. 2014. Quantifying and mapping ecosystem services: Demand and supply of pollination in the European Union. *Ecological Indicators,* 36**,** 131-141.

SEN, A. 1981. *Poverty and Famines - An Essay on Entitlement and Depreviation,* Oxford, Oxford University Press.

SEPA. 2016. Natural Flood Management Handbook. Stirling, Scotland: Scottish Environment Protection Agency.

SEPPELT, R., DORMANN, C. F., EPPINK, F. V., LAUTENBACH, S. & SCHMIDT, S. 2011. A quantitative review of ecosystem service studies: approaches, shortcomings and the road ahead. *Journal of Applied Ecology,* 48**,** 630-636.

SIEW, T. F. & DÖLL, P. 2012. Transdisciplinary research for supporting the integration of ecosystem services into land and water management in the Tarim River Basin, Xinjiang, China. *Journal of Arid Land,* 4**,** 196-210.

SILVERTOWN, J. 2015. Have Ecosystem Services Been Oversold? *Trends in Ecology & Evolution,* 30**,** 641-648.

SMITH, G. 2003. *Deliberative Democracy and the Environment.,* London & New York, Routledge.

SMITH, K. M., LOH, E. H., ROSTAL, M. K., ZAMBRANA-TORRELIO, C. M., MENDIOLA, L. & DASZAK, P. 2013. Pathogens, pests, and economics: Drivers of honey bee colony declines and losses. *EcoHealth,* 10**,** 434-445.

SNH. 2016. Protected Areas. Scottish Natural Heritage. Retrieved from: http://www.snh.gov.uk/protecting-scotlands-nature/protected-areas/protected-areas-az/ [accessed November 2016]

SPASH, C. L. 2009. The New Environmental Pragmatists, Pluralism and Sustainability. *Environmental Values,* 18**,** 253-256.

SUNDSETH, K. & CREED, P. 2008. *Natura 2000: protecting Europe's biodiversity*, European Commission, Directorate General for the Environment Brussels.

TEEB 2010. The Economics of Ecosystems and Biodiversity: Mainstreaming the Economics of Nature: A synthesis of the approach, conclusions and recommendations of TEEB.

TRUCOST & TEEB 2013. Natural Capital at Risk: The Top 100 Externalities of Business. London: Trucost Plc and TEEB for Business Coalition.

TURNER, R. K., BURGESS, D., HADLEY, D., COOMBES, E. & JACKSON, N. 2007. A cost-benefit appraisal of coastal managed realignment policy. *Global Environmental Change,* 17**,** 397-407.

VAN DER SLUIJS, J. P., SIMON-DELSO, N., GOULSON, D., MAXIM, L., BONMATIN, J. M. & BELZUNCES, L. P. 2013. Neonicotinoids, bee disorders and the sustainability of pollinator services. *Current Opinion in Environmental Sustainability,* 5**,** 293-305.

VAN HECKEN, G. & BASTIAENSEN, J. 2010. Payments for ecosystem services in Nicaragua: Do market-based approaches work? *Development and Change,* 41**,** 421-444.

VAN HOOF, L. & VAN TATENHOVE, J. 2009. EU marine policy on the move: The tension between fisheries and maritime policy. *Marine Policy,* 33**,** 726-732.

VATN, A. 2014. Markets in environmental governance — From theory to practice. *Ecological Economics,* 105**,** 97-105.

WALLACE, H., POLLACK, M. A. & YOUNG, A. R. (eds.) 2010. *Policy-Making in the European Union,* Oxford: Oxford University Press.

WEBER, M. 1904/1949. "Objectivity in Social Science and Social Policy" in *The Methodology of the Social Sciences*, E. A. Shils and H. A. Finch (ed. and trans.), New York: Free Press

WILLEMEN, L., DRAKOU, E. G., DUNBAR, M. B., MAYAUX, P. & EGOH, B. N. 2013. Safeguarding ecosystem services and livelihoods: Understanding the impact of conservation strategies on benefit flows to society. *Ecosystem Services,* 4**,** 95-103.

WILSON, J.A. 2002. in The Drama of the Commons, Committee on the Human Dimensions of Global Change, National Research Council, E. Ostrom et al., Eds. (National Academy Press, Washington, DC, 2002), pp. 327–360.

WOODCOCK, B. A., ISAAC, N. J. B., BULLOCK, J. M., ROY, D. B., GARTHWAITE, D. G., CROWE, A. & PYWELL, R. F. 2016. Impacts of neonicotinoid use on long-term population changes in wild bees in England. *Nature Communications,* 7**,** 12459.

YOUNG, O. R. 2002. *The Institutional Dimensions of Environmental Change: Fit, Interplay, and Scale,* Cambridge, MA, MIT Press.

YOUNG, O. R. 2013. *On Environmental Governance,* Boulder, CO, Pradigm Publishers.

ZHU, X., LINHAM, M. M. & NICHOLLS, R. J. 2010. *Technologies for climate change adaptation-Coastal erosion and flooding*, Danmarks Tekniske Universitet, Risø Nationallaboratoriet for Bæredygtig Energi.