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Abstract

The paper gives an overview of some of the large, shallow, semi-enclosed coastal systems (SECS) in Europe, These SECS are important both from the ecological and the economic perspective (socioecological systems) and provide many valuable ecosystem goods and services. Although some of the systems are transitional waters under theWater Framework Directive, this is not the case for all of the systems. The paper adopts a Driver-Pressure-State-Impact-Response approach to analyse the ecological status, vulnerability and future perspectives of these systems in the context of global change.

1. Introduction and scope, definitions and characteristics of shallow, semi-enclosed coastal systems (SECS)

1.1. Scope

This overview addresses some of the largest European shallow, semi-enclosed coastal systems (SECS), including lagoons and transitional waters (TW). These SECS are not only important ecological systems, but also have considerable historical and socio-economic value (Lassere, 1979). In recent years, there has been increasing recognition of the economic importance of SECS through their provision of ecosystem services, although these services are increasingly threatened as SECS are among the most vulnerable coastal systems to both natural and human pressure (Eisenreich, 2005). The overview of SECS is presented from the perspective of the Driver-Pressure-State-Impact-Response adaptive management framework that links human drivers and pressures through to ecological state and the impact on human welfare, and then on to societal responses. The results are interpreted in the context of European environmental legislation, such as the Water Framework Directive (WFD) (EU, 2000) and from the perspective of global change and coastal vulnerability.

Transitional waters include a wide range of typologies including some lagoons (Basset et al., 2006). The WFD does not include a definition of lagoons, but the definition of TW specifies a salinity gradient and significant freshwater inputs. This means that some SECS, such as the Ria Formosa and the Mar Menor, are considered to be sheltered coastal waters (CW) rather than TW, with respect to the WFD. The "legal" classification of these systems as TW or as CW has important implications in the assessment of ecological status, because the type-specific reference conditions for TW are naturally very different from those for CW.

Although many of the SECS are European coastal lagoons, they are not fully representative of the heterogeneity of lagoons, because small eu-haline and hyper-haline lagoons and a myriad of small systems are not included. Most of the systems described can also be considered as coastal wetlands (Perillo et al., 2009). In addition, similar but deeper systems, such as the Gulf of Riga and Puck's Bay, are also considered for purposes of comparison; as well as systems that are not strictly lagoons, such as the Wadden Sea.

The geomorphology of SECS makes them particularly vulnerable to global changes, such as sea-level rise, increased temperatures, storminess, droughts, floods and changes in sediment dynamics. They are "hotspots" of global change and vulnerability to environmental, economic and social pressures, especially when they are associated with river-mouths systems (Newton et al., 2012). Human activities cause changes in demographics, urbanisation, agriculture and land-use, as well as industrial development and shipping that affect the structure and function of these vulnerable and valuable coastal ecosystems.

The overview includes a range of coastal systems, so this section explores the definitions that have been applied to these systems. There is a gradient between the open sea, semi-enclosed bays, coastal lagoons and transitional waters. The United Nations glossary of environment statistics (UNSD, 2006) defines coastal lagoons as "*Sea-water bodies situated at the coast, but separated from the sea by land spits or similar land features. Coastal lagoons are open to the sea in restricted spaces*". Most of the systems included in this overview fit this definition. Coastal lagoons are therefore regions of restricted exchange. However, this is also the case for some enclosed bays and some fjords that may be much deeper (Tett et al., 2003; Newton and Icely, 2006). Geomorphology, in particular the length of the width of marine entrances at high tide relative to the total length of the enclosing barriers, has been used to distinguish enclosed bays from lagoons (Lassere, 1979). This allows for some quantitative interpretation, rather than qualitative, especially with respect to the size of the connection to the adjacent sea. Nevertheless, the degree of connectivity with the sea depends not only on geomorphology and openness, but also on tidal amplitude and the general hydrological regime.

Coastal lagoons are often sub-divided into "choked", "restricted", "leaky" (Kjerfve, 1994) and even "open" (Lassere, 1979; Bird, 1994) with respect to the characteristics of their hydrodynamic exchange properties with the adjacent open sea. The Wadden Sea (southern part of the North Sea) and its extensive intertidal areas behind the Frisian barrier islands has been classified as an "**open lagoon**" (Lassere, 1979; Bird, 1994). **Leaky lagoons** are connected by many entrances to the adjacent sea and are therefore characterised by almost unimpaired water exchange (Kjerfve, 1986). However, **choked lagoons** are connected to the sea by a single or few narrow and shallow entrances, resulting in delayed and dampened tidal oscillation or low water exchange with the open sea. The repletion coefficient of a system (Wolanski, 2007) gives a more quantitative definition based on residence time and tidal prism, but these data are not always available and hence a more arbitrary and qualitative classification is more frequently used.

Table 1 lists some examples of SECS, and similar systems for comparison, and classifies them qualitatively as open, leaky, restricted or choked. It also specifies how they are classified according to the WFD, either as TW or CW.

1.2. Formation and development of SECS

Sediment transport processes are the main mechanism of SECS formation. Barriers of sediment are deposited parallel to the coastline and maybe fragmented into islands and peninsulas. Six main factors influencing the distribution and dynamics of coastal lagoons are: antecedent geomorphology; material characteristics; sediment supply; tectonics; tide range; and climate (Bird, 1994).

The development of SECS is favoured by a low-lying coastline with a fractured geomorphology. Material for barrier formation is sediment drifting along the shoreline. SECS formation also requires sediments of a medium grain size originating either from cliff erosion, riverine input of terrestrial origin, or input from the adjacent seafloor. The evolution of SECS depends mainly on the balance between sediment import and export. The fate of the system is also influenced by: (1) the strength of tidal currents; (2) the force of the wind; as well as (3) the rate of sediment supply. The climate, especially the precipitation, storminess and storm surges, also affects the sediment fluxes, overtopping events and erosion. With respect to tectonics, subsidence potentially leads to an increase in size and depth of the SECS and may even result in reopening of the coastal inlets whereas the opposite will occur in uplift areas (for a detailed review see: Bird, 1994 and references therein).

Table 1
Classification and characteristics of some European SECS (semi-enclosed coastal systems). C = Choked; R = Restricted; L = Leaky; O = Open; TW = Transitional Water; CW = Coastal Water *sensu* WFD-Water Framework Directive; L:C = Ratio of lagoon area to catchment area for some of the lagoons.

Lagoon names(s)	Adjacent Sea/Ocean	Country (ies) (EU abbreviation)			L:C
Bassin d'Arcachon	Atlantic Ocean	FR	R	TW	
Curonian	Baltic Sea	LT, RU	C	TW	0.016
Darss-Zingst Boddenkette	Baltic Sea	DE	C	TW	
Etang de Thau	Mediterranean Gulf of Lyons	FR	C	TW	
Gulf of Riga	Baltic Sea	LT, EE	Bay	C	
Logarou	Amvrakikos Gulf – Ionian Sea	GR	O R	CW & TW	0.007
Mar Menor	Mediterranean	SP	C	CW	0.12
Nordrügensche Bodden	Baltic Sea	DE	L	TW	
Oder/Odra, Szczecin lagoon	Baltic Sea	PL, DE	R	TW	
Papas	Ionian Sea	GR	R	TW	
Puck Bay	Baltic Sea	PL	O	CW	
Ria Formosa	Atlantic Ocean	PT	L	CW	0.19
Ringkøbing Fjord	North Sea	DK	C	TW	0.08
Sacca di Goro	Mediterranean, Adriatic	IT	R	TW	0.03
Salzhaff	Baltic Sea	DE	C	TW	0.29
Venice	Mediterranean Sea, North Adriatic Sea	IT	L	TW	
Vistula	Baltic Sea	PL, RU	C	TW	
Wadden Sea	North Sea	DK, DE, NL	O	CW & TW	
Wismar Bight	Baltic Sea	DE	O		

1.3. Salinity and hydrography

The climate, especially the precipitation conditions, also determines the salinity regime of SECS, which is important for the ecological conditions. The salinity regime is important in defining transitional waters. These are defined as "*shallow aquatic environments located in the transitional zone between terrestrial and marine ecosystems, which span from freshwater to hyper-saline conditions depending on the water balance*" (Kjerfve, 1986). However, the WFD defines TW as "*bodies of surface water in the vicinity of river mouths which are partly saline in character as a result of their proximity to coastal waters but which are substantially influenced by freshwater flows*" (EU, 2000). Some European lagoons have no significant freshwater inputs, and are thus considered to be sheltered, coastal waters in the context of the WFD, whereas they are considered to be coastal lagoons and therefore transitional in character by the scientific community. This is one of the complications for the implementation of the WFD that arises from the characterisation of coastal lagoon systems (Pérez-Ruzafa and Marcos, 2008). In the Atlantic, the existence of significant tidal exchanges may dominate the hydrography and the freshwater inputs may be relatively important. In the Mediterranean, the large meteorological variability of precipitation and thus of freshwater runoff (from drought to flood conditions) creates a very dynamic situation making it particularly difficult to characterise transitional waters. In the Baltic Sea, the low salinity gradients have also prevented the TW classification in some cases.

In principle, all these systems should be considered as TW systems. However, in practice this definition has created several problems (McLusky and Elliott, 2007). There are two major consequences to this problem of system definition. First, TW are considered to be naturally eutrophic in comparison to CW, so the type-specific reference conditions under the WFD are not the same. Second, fish are one of the biological quality elements (BQE) required in the WFD for monitoring TW, but are not a required BQE for monitoring coastal waters.

2. Ecological and economic importance of SECS

SECS are ecologically valuable systems and their ecosystem functions provide valuable ecosystem services. These goods and services are not only economically valuable but also have societal, heritage, aesthetic and scientific values.

2.1. Ecological importance of SECS

Many similarities exist among different European SECS, however there are also the peculiarities exhibited by each system. The relative importance of one particular environmental process with respect to the others is site specific. Important variations include tidal regimes and the flood dominance in different areas of the intertidal zone and the presence or absence of some components of the ecosystem. As they are usually aligned along the coast rather than perpendicular to it, SECS act as barriers for terrestrially derived matter destined for the sea. SECS are complex ecosystems characterised by a natural high spatial and temporal variability and high productivity. They support a rich indigenous fauna and flora because they are sheltered and, in most of the cases, shallow water systems of high productivity. SECS have considerably elevated organic matter concentrations relative to adjacent water masses. In southern Europe, their sheltered nature may also lead to higher water temperatures than the open sea or ocean, to which they are connected. SECS are also spawning and nursery grounds for migratory organisms of ecological as well as economic importance (UNESCO, 1980). SECS are important feeding and nesting sites for a multitude of bird species, as well as important stop-over sites for bird migration or over-wintering sites. Knowledge about SECS ecosystem function is still fragmentary despite the large effort put into research. The main reason for this is the heterogeneity of SECS, restricting cross-applicability of results from one system to another. Table 2 gives examples of SECS that are National Parks, Ramsar sites, Natura 2000 sites and Special Bird Habitats. This table also shows the deterioration of key habitats in some SECS.

2.2. Economic importance of SECS: goods and services provided

The range of ecosystem services provided by SECS is extensive and includes **provisioning** services, also known as ecosystem goods, such as fish and shellfish; **regulating** services, such as water purification; **supporting** services, such as oxygen production from photosynthesis; and **cultural** services, such as recreation and ecotourism (Millennium Ecosystem Assessment, 2005). The ecosystem functions of SECS provide essential services including decomposition, nutrient cycling, and nutrient production. They also function in the regulation of fluxes of nutrients, water, particles, and organisms to and from land, rivers, and the ocean. SECS serve as buffers, sinks and transformers, *e.g.* due to processes like sedimentation, transformation or de-nitrification. Riverine nutrients are retained and SECS thus help to protect the adjacent sea from eutrophication and pollution.

A global overview (UNESCO, 1980) reported a total of 181 programmes specifically dealing with SECS that are coastal lagoons. All of these included the exploitation of lagoon resources demonstrating that coastal lagoons are valuable ecosystems that provide many ecological services. For example, several southern European lagoons have the particularity of being subject to strong anthropogenic pressures due to tourism and/or heavy shellfish/fish

Table 2
Semi-enclosed coastal systems (SECS) that are: National or Regional Parks (P); Ramsar sites (R); Natura 2000 sites (N) and Special Bird Habitats (B), ICZM demo site (D) and Water Framework Directive intercalibration site (I). The table also indicates loss of habitat types $\Downarrow$.

Lagoon	Park	Ramsar site	Natura site	Special bird habitat	ICZM demo site	WFD intercalibration	Sea-grass meadows	Salt marsh	Sand dune	Forested areas	Riparian vegetation
Bassin d'Arcachon			N	B							
Curonian	P	R	N	B	D						
Darss-Zingst Boddenkette	P		N	B							
Etang de Thau			N	B		I					
Logarou		R	N	B			$\Downarrow$				$\Downarrow$
Mar Menor		R		B			$\Downarrow$	$\Downarrow$	$\Downarrow$		
Nordrügensche Bodden	P		N	B							
Oder/Odra			N	B							
Papas			N			I	$\Downarrow$			$\Downarrow$	$\Downarrow$
Ria Formosa	P	R	N	B	D	I	$\Downarrow$	$\Downarrow$	$\Downarrow$	$\Downarrow$	$\Downarrow$
Ringkøbing Fjord		R	N	B			$\Downarrow$				$\Downarrow$
Sacca di Goro	P	R	N	B			$\Downarrow$	$\Downarrow$	$\Downarrow$		$\Downarrow$
Venice		R	N	B			$\Downarrow$	$\Downarrow$	$\Downarrow$		$\Downarrow$
Wadden Sea	P	R		B			$\Downarrow$	$\Downarrow$			

farming (DITTY Project, 2002). However, ongoing eutrophication, socio-economic transformation processes and climate change are a threat to the future structure and function of SECS since these ecosystems are very vulnerable.

SECS are characterised by large fluctuations in the physical and chemical parameters, and, over the last decades, have also shown an enormous potential for residential, tourism, economic (mainly, shellfish/fish farming), and recreational development. Building a network of cause-effect relationships between the different human actions and hydrographical and ecological processes has been proposed as a useful tool for the management of human activities in coastal lagoons (Pérez-Ruzafa and Marcos, 2005).

Table 3 gives a summary of some of the more important economic activities, drivers and sectors in European SECS. It also demonstrates that these areas provide multiple economic activities, adding to their value.

Worldwide, ecosystem services (Woodward and Wui, 2001) in SECS are threatened by: (1) land reclamation, *e.g.* draining of wetlands that are important denitrifying zones (Murray et al., 2006); (2) loss of coastal vegetation (Ehrenfeld, 2000) such as salt marshes that trap carbon and nutrients; (3) loss of habitats such as seagrasses that provide nurseries for juveniles (Beck et al., 2001).

3. Distribution of SECS in Europe

SECS are important physiographic formations of the European coastline and constitute transition zones between terrestrial, freshwater and marine interfaces. In Europe, SECS occupy about 5.3% of the coastline (Barnes, 1980). The SECS have many common features, although they cover a wide geographical distribution from the Baltic to the Black Sea and, as such, are subjected to different environmental conditions and pressures.

3.1. Baltic Sea SECS

SECS can be found in the Baltic along the southern coast, where a long stretch from Germany up to Russia is characterised by a pattern of eroding cliffs of Holocene material and deposition areas rich in coastal lagoons. Tidal range decreases after the entrance to the Baltic Sea and becomes undetectable in the Arkona Basin. Nevertheless, despite being microtidal, wind driven water-level changes, in combination with resonance phenomena, lead to irregular but frequent water exchanges between the Baltic Sea and its SECS. Furthermore, 10–20% of the waves are greater than the 2.4 m European average. The eastern and northern part of the Baltic coast is an area of uplift, whereas subsidence has led to the formation of fjord-like systems along the Jutland and Schleswig-Holstein coast.

Some of the SECS, especially the Nordrügensche Bodden and the Darss-Zingst Boddenkette are interconnected complex systems, whereas the Salzhaff, the Vistula lagoon, and the Curonian lagoon are classical "choked" systems with a rather simple morphology. In addition, several smaller SECS exist along the coast of Holstein (*e.g.* Burger Binnensee on the island of Fehmarn) and the Danish Isles (*e.g.* Odense Fjord on Fyn), whereas along the eastern Polish coast sediment deposition has cut off several former SECS and converted them to coastal lakes. Beside coastal lagoons, the southern Baltic coast is rich in systems that are semi-enclosed bays, *e.g.* Puck Bay (Poland) and the Wismar Bight (Germany). However, there are also numerous semi-enclosed coastal inlets that are formed by other processes than barrier formation by sediment transport. The largest of these systems is the Gulf of Riga that is enclosed by the islands of Saarema and Muhu. The remaining connection is less than 20% of the stretch covered by these islands but it is a relatively deep system and therefore a semi-enclosed bay rather than a SECS. Other examples are the numerous flads of the Finnish Archipelago coast. These systems, still having substantial water exchange with the adjacent sea, are formed by land uplift of former bays. However, their principal hydrological conditions are similar to coastal lagoons formed by sediment barriers. Schiewer (2007) gives a comprehensive survey of the ecology of the coastal lagoons and coastal inlets of the Baltic Sea. There are no invertebrates of harvestable size living in the Baltic SECS, so there is little shellfish harvesting activity.

3.2. Atlantic and North Sea SECS

There are a number of large SECS along the Atlantic coasts of Europe, especially in France, *e.g.* the Bassin d'Arcachon; in Portugal, *e.g.* the Ria de Aveiro and the Ria Formosa; and in Spain, *e.g.* the Marismas de Odiel. The use of the term "Ria" in Portugal differs from its use in Galicia, where they are drowned river-valleys, therefore the Galician Rias are not included in the overview of SECS. The eastern North Sea is fringed by the largest European SECS, the Wadden Sea.

The Arcachon Basin is a triangular meso-tidal coastal lagoon of 156 km^2 in SW France on the Atlantic coast (de Wit et al., 2005). It includes a delta and a long dune system. It is connected to the

Table 3
Major economic activities and drivers in European SECS (semi-enclosed coastal systems).

Lagoon	Agriculture	Golf	Damming of streams	Aquaculture ponds	Fishing	Tourism and ecotourism	Watersports	Shipping, ports and marinas	Inlet consolidation	Dredging and sand extraction	Salt extraction and salt pans	Land reclamation	Urbanisation	Industrial development
Bassin d'Arcachon	X			X	X	X	X	X		X				
Curonian	X		X		X	X	X	X				X	X	X
Gulf of Riga	X				X	X	X	X						
Logarou	X		X	X	X									
Mar Menor	X	X			X	X	X	X	X	X	X	X	X	
Papas	X		X	X										
Ria Formosa	X	X	X	X	X	X	X	X	X	X	X	X	X	
Ringkøbing Fjord	X		X	X	X	X	X	X		X		X	X	X
Sacca di Goro	X	X		X	X	X					X	X		
Oder- Szczecin	X				X	X	X	X		X				
Thau	X			X	X	X	X	X	X				X	X
Venice	X			X	X	X	X	X	X	X		X	X	X
Vistula	X				X	X	X	X						
Wadden Sea	X			X	X	X	X	X		X				X

Atlantic through several inlets, through which there is an important tidal exchange. The intertidal area is about 140 km^2 and the tidal currents are about 2 $m\ s^{-1}$. These tidal currents also move the sand banks in the lagoon and the position of the inlets. The lagoon is estuarine as it receives the waters from several rivers and also runoff from the land, and thus also qualifies as a TW under the WFD. The climate is temperate, although there can be violent storms causing widespread destruction, such as cyclone Xynthia (March 2010). The rich ecology of the lagoon supports abundant birdlife. Economic activities in the lagoon include aquaculture of the oyster *Crassostrea angulata*, which covers about 1800 ha of the lagoon and produces about 18,000 tonnes per annum. There is also some fish farming, but this is insignificant in comparison to the oysters. Fishing, boating and tourism are other important economic and leisure activities, as there are some well-known coastal resorts along the lagoon.

There are a multitude of small systems called “lagoas” along the coast of Portugal and two large semi-enclosed systems, the Ria de Aveiro and the Ria Formosa. This is another example of confusing nomenclature. The Ria de Aveiro has large freshwater inputs from the Rio Ave and a number of smaller rivers, therefore it is classified as a TW under the WFD. However, the Ria Formosa has only one small permanent river that flows into the east of the lagoon, the Rio Gilão. Fourteen small, torrential streams flow into the lagoon after heavy rainfall. The Ribeira de São Lourenço stream, which used to flow into the west of the lagoon, was completely damned in the 1800’s with a dyke (Newton and Icely, 2002). As a consequence, there is no significant freshwater input to the Ria Formosa coastal lagoon and so it is not classified as a TW but instead as “sheltered CW” in the context of the WFD typology.

The Wadden Sea is the largest SECS in the North Sea, stretching 500 km from the island of Texel in the Netherlands to north of Esbjerg in Denmark. This trans-boundary system is separated from the North Sea by a string of islands, the Dutch Frisians and the Danish Jutland islands. The Wadden Sea is shallow and its 10,000 km^2 area includes large expanses of tidal flats and wetlands that support a rich and diverse fauna, including seals and huge numbers of birds (Colijn and van Beusekom, 2005). The Wadden Sea is protected by a string of National Parks as well as being a Ramsar and a UNESCO World Heritage site, thereby attracting large numbers of tourists. Nuisance blooms of *Phaeocystis* frequently affect the use of the shore.

Also on the North Sea but on a smaller scale, Ringkøbing Fjord covers an area of almost 300 km^2 (Petersen et al., 2008). It is not a drowned glacial river valley, such as typical fjords, but a coastal lagoon, also illustrating the varied nomenclature applied to SECS. It is very shallow with an average depth of 2 m, and has a water residence time of 3–4 months. The catchment area of this lagoon is 3500 km^2 and land use is dominated (60%) by intensive agricultural activities especially animal husbandry. The lagoon has experienced a regime shift in recent years (Hakanson and Bryhn, 2008).

3.3. Mediterranean European SECS

The study of coastal lagoons has a long tradition in the Mediterranean (see for example Petit, 1953, 1962; Mars, 1966; Lasserre and Postma, 1982; Carrada et al., 1988, and the works in the committee of “étangs salés et lagunes” of the *Commission Internationale pour l’exploration scientifique de la Mer Méditerranée*). However, the information is dispersed, heterogeneous and the nomenclature often confusing. In the Mediterranean, there are more than 50 coastal lagoons with some hydrological or ecological data published in scientific literature (Pérez-Ruzafa et al., 2011). Sabetta et al. (2007) list 26 coastal lagoons only for Italian waters, excluding Sardinia. In Greece, there are at least 40 coastal lagoons under aquaculture exploitation (Schmidt and Spagnolo, 1985). There are also many coastal lagoons on the non-European shores of the Mediterranean. From so many systems in the Mediterranean, only 3 large European lagoons with abundant data are described as examples: the Mar Menor (Spain), the Etang de Thau (France) and the Lagoon of Venice (Italy).

The Mar Menor is a large SECS on the Mediterranean coast of Spain. It is 21 km long and 135 km^2 but only 7 m at its deepest point. The shores of the lagoon have been inhabited for centuries, with records dating back to Phoenician and Arab settlements. Nevertheless, in the 20th century it was used as a dumping site for mine tailings, whereas now it is valued as an important tourist resort. The Mar Menor includes several protected sites, with important sea-grass meadows and a rich birdlife. However, in recent years fisheries have declined and the jellyfishes *Rhizostoma pulmo* and *Cotylorhiza tuberculata* have proliferated, which affects the use of the beaches by swimmers (Pérez-Ruzafa et al., 2002). Fisheries are mainly for marine migrant fishes that use the lagoon as nursery areas and feeding grounds. The main commercial species are fishes from the Families Sparidae, Mugilidae, Anguillidae and Moronidae (Kapetsky and Lasserre, 1984), which are present in more than the 75% of the Mediterranean lagoons (Pérez-Ruzafa et al., 2007b), but shrimps and clams can be locally very important.

The microtidal Etang de Thau (de Wit et al., 2011) is the largest of a string of coastal lagoons fringing the Mediterranean coast of France. The area of the lagoon is about 7500 ha and it has an average depth of 5 m. The deepest part of the lagoon is 32 m. The volume of the lagoon is about 340 million m^3 but the tidal exchange, between 0.75 and 3.7 million m^{-3} d^{-1}, is far smaller than in the Bassin d'Arcachon on the Atlantic coast. There are important freshwater inputs from streams, runoff and groundwater inputs. The climate is Mediterranean and the salinity increases during the dry, summer months. The Thau lagoon supports a rich ecology and is well-known for its abundant avifauna. Aquaculture of oysters and mussels is an important economic activity. There are about 600 shellfish farms employing about 2000 people and producing about 12,000 tonnes of oysters per annum. Other important activities include fishing, sailing and navigation. The lagoon is linked to the Canal du Midi and is therefore an important navigation route.

The Venice Lagoon (Lasserre and Marzolla, 2000; Solidoro et al., 2010) is the largest lagoon of the Mediterranean Sea (Upper Adriatic) with a surface area of about 550 km^2, of which 418 km^2 are intertidal (the largest tidal range in the Mediterranean). It is connected to the northern Adriatic Sea through three inlets: Lido, Malamocco and Chioggia. The Venice Lagoon supports different kinds of ecosystems, including dunes, tidal channels, bare mudflats, sea-grass beds and salt marshes. Like all others wetlands, it has an important function in animal and plant interaction, routes for animal migration, plant dispersal (including seeds) and maintenance of biodiversity. The extensive wetlands include a salt-water lagoon and shallow water ponds that are sub-divided by numerous natural and artificial canals. The historical town of Venice is located on an island in the centre of the lagoon and the city is frequently flooded when tides and storm surges result in an "Aqua Alta" event.

3.4. Black Sea SECS

The Black Sea is an inland sea bounded by a coast shared by 6 countries: Ukraine, the Russian Federation, Georgia, Turkey, Bulgaria, and Romania. The northwestern coast of the Crimea is low-lying and abounds in coastal lakes, that are superficially separated from the sea, although there may be groundwater exchanges. There are also many coastal lagoons, called "limans" that stretch north from the Danube to the mouth of the Dnieper. Liman is the term used for lagoons found along the western and northern coast of the Black Sea, as well as along the lowest part of the Danube. Examples of limans include Lake Varna in Bulgaria, Lake Razelm in Romania and the Dniester liman in the Ukraine. The largest limans are those of the Dniester, Bug, and Dnieper rivers, and they are connected with the sea. Smaller limans lose so much water by evaporation that they form closed saline lakes (Kubijovyč and Teslia, 1984–1993).

4. Linking of river basins districts and catchments to SECS

Many processes and pollution in the SECS originate in the catchment, including rivers and their drainage basins. SECS water quality management therefore requires the management of nutrient and pollutant load in the river basin. The EU WFD links rivers, TW and adjacent CW to river basin districts that are to be managed as a single unit. The impact of a river depends on the water discharge and the volume of the TW, CW or adjacent sea. The water exchange time is a reasonable indicator for the impact of river discharges. High river discharges are usually linked to high nutrient and pollutant loads. The River Basin Management Plans and the Programme of Measures represent the policy response of Member States to implement all necessary measures to prevent deterioration and to restore all water bodies to good ecological status within 15 years of the entry into force of the WFD (Euro-Site-Manager, 2012). The emissions and discharges of priority hazardous substances will be phased out and eventually cease, which should progressively reduce pollution. Member States must undertake Risk Assessments to assess the status of water bodies in the future. This task is linked to the analysis of pressures and impacts and its objective is to identify water bodies at risk of not achieving the WFD objective and breaching the "prevent deterioration" principle.

In the wet climate of northern Europe, the river discharge and nutrient loads have a much higher impact on SECS compared to the dry Mediterranean region. In the Baltic region (river basin 1.74×10^6 km^2; average river discharge 14,000 m^3 s^{-1}) 254,000 m^3 km^{-2} y^{-1} are discharged by rivers. In the Black Sea region (1.87 x 10^6 km^2; 11,100 m^3 s^{-1}) about 187,000 m^3 km^{-2} a^{-1} and in the Mediterranean drainage basin the discharge is in the range of only 50,000 m^3 km^{-2} y^{-1}. This is also true for Atlantic lagoons in S. Europe, for example, the Ria Formosa on the south coast of the Algarve has few freshwater inputs, only one is permanent and the others are torrential.

4.1. Trans-boundary issues

Several European SECS are shared between two countries, and many more have shared watersheds. The tri-lateral Wadden Sea plan is an example of how trans-boundary cooperation between three nations can work for the management of SECS. Since the 1980s, dramatic political, social, economic and natural changes and challenges have affected the management of SECS in Europe. After the fall of the Iron Curtain, political and social changes have been rapid in many of the new EU member states. The EU-membership accelerated change, not only because of the new agriculture and industry policies, but also the implementation of new standards. The Curonian lagoon used to be part of the coast of the Soviet Union, but now it is divided between Lithuania, a member state of the EU, and Russia. The Oder/Odra lagoon at the German/Polish border has been chosen for the following more detailed discussion. The German part of the Odra region belonged to the former German Democratic Republic. Like eastern Germany, Poland was subject to social changes and its transitional economy is still facing rapid changes. During the last decade, the economic and social developments in Germany and Poland were largely independent and caused strong social and economic gradients. Social problems are increasing the gap between the flourishing seaside resorts and the hinterland. Cross-border cooperation as well as competition will increase and cause social and economic transformations. Furthermore, many legal challenges are taking place.

5. DPSIR framework and European SECS

This overview uses a conceptual framework in which environmental problems affecting coastal lagoons may be assessed by the evaluation of driving forces (D), pressures (P), states (S), impacts (I) and policy responses (R). The DPSIR methodology is based on an OECD-Organization for Economic Cooperation and Development concept (OECD, 1994). It has also been further developed and widely applied by projects such as LOICZ-Land ocean interactions in the Coastal Zone and EU projects such as DITTY, as well as organisations such as the EEA-European Environment Agency and UNEP-United Nations Environment programme. The DPSIR framework can be used to assess a wide range of other pressures, conditions and scenarios pertinent to the application of EU Directives (Zaldivar et al., 2008). In recent years, there has been a gradual change in the use of the DPSIR terminology. State is now considered to be “state change” and Impact is “impact on human system” (Elliott et al., 2006), and on human welfare. In this way, the framework uses an integrated “system approach” that considers the SECS as a Socio-Ecological System (Newton, 2012).

Eutrophication is a good example of a clear causal chain between the components of DPSIR. Drivers include urbanisation (discharges of domestic sewage and detergents), agriculture (fertilisers and manure) and industry (organic matter from food processing and paper mills). The resulting pressures (inputs of organic matter, nutrient load and enrichment) causes an adverse change in the state of the physico-chemical quality elements (*e.g.* transparency, oxygen conditions, nutrient stoichiometry), and state changes in the biological quality elements (*e.g.* composition, abundance and biomass of phytoplankton and other biota such as macro-invertebrates). Water bodies that are eutrophic due to human induced nutrient enrichment fail to achieve Good Ecological Status. The deteriorating **state** of the system has **impacts** on human welfare and activities, such as the closure of shellfish harvesting, decreasing fisheries catches and loss of tourism revenue. Member States must respond by implementing management measures and **responses** such as upgrading sewage treatment plants, improving oxygenation by using fountains for example, or restoring fringing, riparian vegetation.

5.1. *Drivers of change in European SECS*

SECS are complex ecosystems balancing ecological processes and human activities. The multiple activities and sectors are **drivers** of changes in SECS. European SECS are valuable not only for tourism and recreation but also provide protected areas for harbours and the aquaculture industry. An assessment has shown that fishing, aquaculture and tourism were the most common and valued uses in European lagoons (Seeram, 2008).

Table 3 gives a summary of some of the more important economic activities in European SECS. It also demonstrates that these areas provide multiple economic activities, adding to their value.

In general, Mediterranean lagoons support a wide range of human activities (**drivers**) including urban development and land reclamation, salt mining, harbours, tourism and nautical activities, fisheries and aquaculture. Most of these activities can be concentrated in a given lagoon, as the Mar Menor (Pérez-Ruzafa and Marcos, 2005).

Agricultural practices, food processing and urbanisation lead to **pressures** (nutrient and organic matter inputs) that cause the eutrophication of SECS. However, eutrophication is not the only important watershed-lagoon interaction. Other sectors, such as industry, introduce effluents that are a contaminants pressure to SECS, *e.g.* metals and persistent organic pollutants, so that the seafood maybe unfit for human consumption because of high concentrations of contaminants.

Construction of coastal defenses and port structures, as well as the dredging of navigational channels for maritime transport are other important **drivers** that change hydro-morphology and lead to **pressures** on SECS. Several fish species, like eel, salmon and trout migrate through lagoons into the river system to spawn and constructed structures, such as dams and dykes, place physical obstacles in the river that hamper the fish migration.

Shipping and ballast water also causes **pressures** by the introduction of anti-foulant contaminants and non-indigenous species (NIS). Linked river-TW-CW systems provide a convenient path for the spreading and migration of species. The degradation of the ecosystems in the river catchment and the TW is a serious threat that has already altered the fauna in many cases. The large number of Natura 2000 sites (see Table 2) in the SECS demonstrates the need for an integrated **response** including nature protection management for river-TW-CW systems.

5.1.1. *Urbanisation and tourism*

European SECS are attractive and productive environments with associated urban and tourism development. SECS are valued recreational and touristic areas because of their shelter and consequent lack of strong wave action. Many areas of coastal wetland have been drained for the construction (**driver**) of urban areas, tourist resort, marinas, golf courses and even airports (*e.g.* Faro, Venice and Barcelona). SECS in Europe are situated along the coastline in highly populated areas. The increasing development of SECS for tourism is resulting in greater **pressure** from domestic effluents and seasonal variability of population and inputs. The more popular touristic lagoons include Venice, Mar Menor and Ria Formosa.

In southern Europe, SECS also play important roles for migratory and wintering birds. The Ria Formosa has become a major contributor to the Algarve’s tourism (**driver**), either by directly attracting bird watchers, or indirectly by supporting restaurants, hotels, car rentals etc. In 2002, tourism from this lagoon contributed €2,585,000 to Portugal’s economy (Serpa et al., 2006). However, runoff from golf courses contributes to high nutrient inputs (**pressure**) into the lagoon and the high number of tourists, particularly during the summer, places increased stress on the sewerage disposal system (Loureiro et al., 2005; Mudge et al., 2007).

The Mar Menor shows a similar situation with 400,000 visitors in 2008, mainly concentrated in the summer season, and a mean inter-annual rate rising close to 10% and maintained during the last decade (Pérez-Ruzafa and Marcos, 2008). Increasing **pressures** originate from golf courses, urban and marina development (**driver**) in the watershed and the Mar Menor coastline.

Venice is an extreme case of imbalance between residents and visitors. Venice has an ageing and decreasing population of only 60,000 residents in the historic centre of Venice, and a total of 400,000 living in the municipalities around the lagoon (CORILA, 2008). The local residents are moving out of the city because of the decline in the provision of basic services. Costs and taxes are increasing due to the strain on infrastructure from the overwhelming numbers of visitors. Venice has an annual influx of 2 million tourists (**driver**). The economy is dominated by the tourism sector, which provides the main income for Venice. However, tourism places stress upon the city infrastructure and **pressures** on the natural system. Problems include an increase in boat transport, pollution, habitat destruction, and waste treatment. The increased pressure of nitrogen (N) and phosphate (P) from untreated sewage of the residents and numerous tourists has caused great changes in the state of the macrophytes and the ecological status (**state**) of the lagoon (Ravera, 2000), but a reversal in the eutrophication trends has been observed in the last decade (Solidoro et al., 2010).

5.1.2. Civil engineering, alterations in hydrography and substrate

The construction of ports, marinas and consequent dredging for navigation alters the geomorphology, sediment transport, sea-floor integrity and hydrodynamics of SECS. There is an excellent historical record of the civil engineering measures adopted in Venice to preserve the equilibrium between the opposing aquatic and terrestrial forces, and so Venice serves as an example for this section: see Lasserre and Marzolla (2000), and references therein.

Venice is a European lagoon that has been intensively urbanised and profoundly modified morphologically. The city was founded in AD 400, and the morphology and water dynamics of the lagoon have been always of great concern, with a growing awareness that the city was sinking. The objective of the multiple interventions has been to control the excessive sedimentation of the canals of Venice, in order to maintain a permanent balance between land and water. In the XIV century, the lagoon of Venice was very different from today. There were large rivers flowing into the lagoon, 5–8 unstable inlets, and a large area of wetland and marshes. The tidal flats were prone to siltation, with a consequent risk of infilling the lagoon. Venice was one of the richest cities of Europe and became a great Mediterranean power. The siltation therefore jeopardised the survival of Venice as a commercial and military power, as well as the physical existence of the lagoon. From the XIV to the XVII century, great care was taken by the *Serenissima Repubblica* to defend the lagoon "*against sea, rivers and man*". During this period, the lagoon was subjected to huge engineering works that diverted most of the rivers to the sea. At the end of the 18th century, the Venetian Republic started the building of the sea-defenses along the coastal strip, called *Murazzi*, that are still in existence.

Present conditions are somehow reversed with respect to the XIV century. Industrial development and the port of Mestre and Porto Maghera were the main **drivers** for various actions such as the construction of long jetties. Port structures were constructed and channels were dredged for navigation. The jetties were built between the late 19th and the early 20th centuries, and changed the hydrography of the three inlets, thereby preventing sediment transport from the sea. River flow into the lagoon was also diverted, so riverine sediment input was reduced and land subsidence increased water depth. In addition, natural phenomena such as sea-level rise and subsidence have led to sinking of the city (about 23 cm in the last century). The cumulative **pressures** caused morphological deterioration and frequent flooding. The increased frequency of high water floods "*Aqua Alta*" has led to a societal **response** in the form of a large engineering project, known as MOSE-MOdulo Sperimentale Elettromeccanico (Experimental Electromechanical Module), to protect the city of Venice from floods. It consists of rows of mobile gates able to isolate the lagoon from the Adriatic Sea when the tide reaches above 110 cm and up to a maximum of 3 m.

5.1.3. Agricultural drivers: crops, animal rearing and aquaculture

Agricultural **drivers** are some of the most important causes of change in European SECS. Many areas of coastal wetland have been drained and converted to agriculture. Fertiliser application in the EU has remained high, despite the Nitrate Directive (EU, 1991b). In addition to the cultivation of food crops, a new agricultural driver is the increase in biofuel-crops because of the high price of energy. Changes in European lifestyles are resulting in an increase in demand for protein. Intensive animal raising such as pig, chicken and poultry farms results in increased **pressures** from the resulting manure and farm effluents.

Another important **driver** is the increase in intensive aquaculture giving rise to **pressures** from the resulting effluents or excessive organic matter from waste feed. In some countries, aquaculture contributes significantly to the national economy and aquaculture is expected to play an important role in the future as a supplement to stagnant fisheries and the increasing demand of marine products for human consumption (Olsen et al., 2008). As a consequence, there is a rapid increase in aquaculture in the SECS, as well as expansion of open ocean aquaculture. Aquaculture has grown rapidly in SECS due to the easy access and the sheltered locations available. Many fish farms and shellfish farms are located in areas with rapid water exchange, since there are large environmental impacts if the farms are placed in sheltered locations with low water exchange (Holmer et al., 2005).

Coastal lagoons are extensively exploited for aquaculture, especially for mollusc farming (Zaldivar, 2006; Melaku Canu et al., 2011). Shellfish farming is one of the main economic activities in SECS. Mollusc farming has undergone rapid expansion over recent decades attaining 13×10^6 tons in 2008, which represents up to 25% of the total aquaculture yield (FAO, 2010). More than 90% of seafood production is of bivalve molluscs, mainly mussels, oysters and clams that are farmed principally in shallow and sheltered SECS. Examples where more than 50% of the SECS is exploited for shellfish farming include Thau, Prévost, Sacca di Goro, some sub-basins in the Venice lagoon and the Ria Formosa. In Portugal, 90% of the bivalve production comes from the Ria Formosa and this industry supports 7000 families.

The shallow depth and low tidal exchange make SECS sensitive to aquaculture **pressures** and are expected to alter both sediment and water quality (Kaiser et al., 1998). However, shellfish aquaculture is assumed to have less **pressure** than fish and crustacean farming, since bivalve molluscs feed on natural phytoplankton and seston and do not require external feed supply (Naylor et al., 2000). In contrast to the culturing of fish, shellfish production is considered as a possible tool to compensate nutrient loading, and shellfishes are in several countries grown to compensate N and P loading, *e.g.* from agriculture (Lindahl et al., 2005; Ferreira et al., 2009). However, above certain biomass densities the mollusc metabolism may exacerbate eutrophication.

The main pressures to SECS from the agricultural drivers are increased inputs of organic matter (manure and aquaculture effluent), fertilisers (N and P) and inputs of contaminants, such as pesticides and herbicides.

5.1.4. Industrial drivers

Many SECS are associated with industrial areas. Striking examples are the industries of Porto Maghera near Venice, the industrial complexes near the Marismas de Odiel in S.W. Spain and the industries centred on Aveiro in Portugal. Power generation can also impact lagoons by altering the temperature regime (thermal pollution) or the hydrological regime. A good example of this is the Etang de Berre, on the Mediterranean coast of France where electricity generation has significantly altered the salinity of the lagoon and consequently the fisheries.

Industrial **drivers** increase the pressure of many different types of effluents to European SECS. Industrial effluents (**pressure**) include a huge variety of different substances and these are mainly addressed in the sections below on metal pollution and organic pollution. The food processing industry and the paper industry (**drivers**) is addressed in the section on eutrophication, because the main **pressures** are nutrient and organic matter pollution.

5.1.5. *Other drivers*

Because the SECS are shallow, fishing boats are usually small and for artisanal fisheries (**driver**). This limits the development of fisheries in SECS, although landings of catch may be quite high. This is because the SECS are good harbours and there are usually fishing ports. Dredged channels often connect these to the adjacent coastal waters where the main fishing activity takes place. So, although the catch maybe landed in a fishing port inside the SECS, the fish were often captured offshore. Olhão, a fishing port in the Ria Formosa is a good example of this. It is the third port in Portugal for landings, but the fish are mainly caught outside the lagoon.

The sheltered SECS provide natural harbours that have frequently been developed into commercial harbours and marinas for maritime transport (**driver**). However, because of the shallow channels and sediment fluxes, hard structures are often constructed to consolidate the inlets and SECS need to be dredged regularly (**pressure**). This is often combined with a mineral extraction sector (**driver**) for the construction industry or beach replenishment.

Another mineral extraction industry (**driver**) is salt extraction. There has been extensive conversion of the wetlands (**pressure)** of SECS to make evaporation ponds since Roman times, particularly in the Mediterranean, but also in some Atlantic SECS, *e.g.* the Ria de Aveiro and the Marismas de Odiel. This may also coincide with chemical industries (**driver**) that use salt, such as highly polluting chlor-alkali plants (**pressure**).

5.2. *Pressures on European SECS*

Many SECS are under the increasing influence of human activity. Because of their economic and social importance, SECS are subjected to severe anthropogenic **pressures** exacerbated due to restricted water exchange with the adjacent coastal waters. Anthropogenic pressures are evident at all scales, from local pollution and interference with natural deposition and sediment transport regimes as well as the global alteration of sea levels due to human-induced climatic change. Coastal zones are under increasing human pressures, which make them very sensitive and vulnerable to perturbations (Crossland et al., 2005) and hotspots of global change (Newton et al., 2012). SECS are often developed as a regional resource and may receive waste materials (**pressures**) from urbanisation and other types of human activities (**drivers**). The pressures can be direct, from commercial and recreational activities within the lagoon, or indirect, primarily through loading with nutrients, organics and contaminants lost from urban areas and industrial and agricultural activities in the catchment area (Nixon, 1995; Conley et al., 2000). Pressures on SECS include the disruption of sediment cycles, the hydrological cycle, biogeochemical cycles, habitat loss and overexploitation of biotic and abiotic resources. The main pressures are an excess of organic matter and nutrient inputs from the watershed, which impact water quality and ecosystem good and services (Nixon, 1995; Cloern, 2001). In conclusion, SECS experience significant pressures, which are expected to increase in the future as the coastal population grows.

5.2.1. *Pressures from effluents*

Urban sewage, industrial effluents, agricultural runoff and aquaculture effluents are characterised by a relatively high concentrations of phosphorous, nitrogen, metals, organic matter, and organic contaminants such as pesticides and herbicides. This complex mixture may have multiple adverse effects: ecologically (**state**), aesthetically and also for public health (**impact**). The contaminants may be retained, accumulated and even concentrated in SECS. The **pressures** from urban waste increased dramatically in Europe up to the 1980s as a result of increasing urban population (Nixon, 1995; Howarth and Marino, 2006; Conley et al., 2007). Table 4 summarises the effluent and runoff into European SECS.

Two systems in Portugal provide an interesting contrast. The Aveiro lagoon was the site of industrial development in the 20th century with some very polluting industries (Abreu et al., 2000) and effluents from chlor-alkali plants. In contrast, the Ria Formosa lies in an agricultural area, now increasingly developed for tourism and golf. Nutrient inputs to the lagoon also come from the surrounding area (Campina de Faro) that is intensively farmed and several golf courses that border the lagoon (Newton et al., 2003). The lagoon is vulnerable to anthropogenic eutrophication because of the local resident population (100,000 inhabitants) that is multiplied by a factor of 4 or 5 in the summer months, when the plant capacity of Urban Waste Water Treatment (UWWT) is overwhelmed.

5.2.1.1. Nutrient and organic matter pollution. After the 2nd world war, the use of synthetic fertiliser increased dramatically leading to the severe eutrophication of lakes and coastal waters from about 1970 (Howarth and Marino, 2006; Conley et al., 2007; Viaroli et al., 2008). Also, reclamation of wetlands, the straightening of channels and the destruction of riperain vegetation has reduced the natural nutrient retention and de-nitrification resulting in increased eutrophication problems in SECS. The time lag between the increase in the use of fertiliser and eutrophication effects demonstrates a large buffer capacity of aquatic ecosystems (Carstensen et al., 2006) and homeostatic mechanisms based on top-down control in the trophic webs of some lagoons.

Eutrophication from agricultural sources was relatively low until the 1950s. In 1995, the European Environment Agency (EEA) report "*Europe's Environment: the Dobris Assessment*" identified eutrophication of inland and marine waters as a European wide

Table 4
Effluent pressures and runoff into European SECS (semi-enclosed coastal systems).

Lagoon name(s)	Rainstorm urban runoff	Agricultural runoff	Golf course runoff	Chicken farms effluent	Pig farms effluent	Aquaculture ponds effluent	Untreated sewage effluent	UWW effluent	Industrial effluent	Food processing effluent
Bassin d'Arcachon	X	X		X	X			X		
Curonian lagoon	X	X		X	X		X		X	
Etang Thau	X	X		X			X	X		
Logarou		X			X		X			
Mar Menor	X	X	X	X			X			
Oder (Odra lagoon)		X					X			
Papas		X				X	X			
Ria Formosa	X	X	X	X	X	X	X	X		X
Ringkøbing Fjord	X	X	X			X	X			
Sacca di Goro		X		X	X	X	X			X
Venice lagoon	X	X			X	X	X		X	

problem of major concern. The EEA (2003) review "*Europe's water: An indicator-based assessment*" stated that progress was being achieved in improving water quality and quantity particularly in the European Union but many of Europe's rivers, lakes, estuaries and coastal waters are still affected by human activities leading to eutrophication.

The main nutrients causing eutrophication problems in lagoons are nitrogen (N) and phosphorus (P) (Vollenweider, 1992). The N load delivered to coastal zones is considered as the main cause of macro-algal blooms and ecosystem perturbations (Valiela et al., 1997; Howarth and Marino, 2006). In nitrate-rich waters, nitrophilic opportunistic species take advantage of their capacity to exploit transient nitrate availability and store nitrate. However, the nitrate storage capacity can be saturated, lowering their growth potential (Naldi and Viaroli, 2002). Table 5 summarises the symptoms of eutrophication in European SECS.

Lagoon water quality and sensitivity to nutrient and organic loadings from the watershed have been assessed mainly with water chemistry and phytoplankton indicators, basically referring to the trophic reference system proposed by Vollenweider and Kerekes (1982) and Nixon (1995). Vollenweider et al. (1998) proposed a trophic index (TRIX) that integrates chlorophyll-*a*, oxygen saturation, total N and total P to characterise the trophic state of coastal marine waters. This index has been applied to coastal lagoons (Coelho et al., 2007; Salas et al., 2008). TRIX is based on the assumption that eutrophication processes are related primarily to the phytoplankton community, which may not be the case for shallow coastal lagoons where both micro-phytobenthos (MPB) and benthic vegetation are important components of the primary producer community. Other indices that consider the coverage of benthic phanerogams and opportunistic macro-algal species are more suitable in these cases (Giordani et al., 2009). Newton et al. (2003) used and compared several indices of eutrophication in the Ria Formosa.

The issues to be analysed when assessing eutrophication are complex and cannot be resolved by considering only simple variables and linear relationships. Nevertheless, a set of basic variables that are indicative of ecosystem properties and functions (**state**) can be easily measured and are suited for classification and assessment of sensitivity to external stressors (Viaroli and Christian, 2003; Viaroli et al., 2004). Basic morphometric parameters, hydrological variables, sediment characteristics and biological elements have been extensively applied and validated in deep aquatic ecosystems. Nevertheless, applications to shallow coastal lagoons require *ad hoc* calibration and further validations using the weight-of-evidence approach. In order to bring together information from multiple indicators, metrics that allow integration or combination of multiple variables will also greatly improve the capacity of representing ecological status or sensitivity to a given stressor (Viaroli et al., 2004). The mean depth is an indicator of the development of the water–sediment interface with respect to the water volume. The ratio of freshwater discharge to recipient volumes could be either a measure of freshwater flushing or a proxy of sensitivity to pressures from watershed. According to Vollenweider (1992), the combination of hydrological variables with loading estimates gives an assessment of the lagoon sensitivity to eutrophication. However, this approach is more appropriate for deep aquatic ecosystems, dominated by phytoplankton communities. Water budgets and nutrient loadings have been widely used for assessing the net ecosystem metabolism (NEM) of coastal lagoons with a wide array of primary producer communities (Giordani et al., 2005, 2008). Basically, NEM is calculated with the LOICZ biogeochemical model from P loadings; thus NEM gives a measure of the trophic status and of its dependency on nutrient delivery from the watershed.

Table 5
Symptoms of eutrophication in European SECS (semi-enclosed coastal systems). P = *Prorocentrum sp.*, A = *Alexandrium catenella*, D = Diarrhetic Shellfish Poisoning.

Lagoon	Large inputs of N	Large inputs of P	Unbalanced N:P and N:Si	Problems with oxygen	Anoxic sediment and H_2S	Release of P from sediments	Phytoplankton blooms and HAB	Opportunistic green algae	Loss of sea-grasses	Decreased benthos biodiversity	Decreased fish biodiversity	Birds community effects	Fish kills
Bassin d'Arcachon	X		X				P	X	X				
Curonian lagoon	X	X	X	X	X		X	X		X			X
Etang de Thau	X		X	X			A	X					
Logarou									X	X	X	X	
Mar Menor	X	X	X				X	X		X	X	X	X
Oder/Odra	X	X	X	X	X	X	X			X	X		X
Papas	X	X		X	X			X	X	X	X		X
Ria Formosa	X	X	X	X	X	X	D	X	X	X	X	X	X
Ringkøbing Fjord	X	X	X	X		X	X	X	X	X		X	
Sacca di Goro	X		X	X	X			X	X	X	X		
Wadden Sea	X		X	X	X	X	X	X	X			X	

The susceptibility to eutrophication is not only a simple function of nutrient loadings, but also depends on sedimentary processes that are mainly controlled by a suite of sedimentological and geochemical variables. Granulometry and sedimentary organic matter provide basic information on sediment composition and are important determinants of exchanges of oxygen and nutrients across the water–sediment interface. The C:N ratio of the sediment organic matter pool is a good indicator of organic matter lability (Enriquez et al., 1993). The sedimentary carbonate content can be used to assess the capacity of the sediment to retain phosphate through the calcium/carbonate/phosphate system (Golterman, 1995a,b; Rozan et al., 2002). Reactive iron provides an indication of the sediment capacity to buffer against sulphides and phosphate (de Wit et al., 2001). Yet, the above biogeochemical variables are also related to benthic vegetation and, to some extent, the macroalgae to sea-grass ratio could be a proxy of their relative dominance.

Sedimentary variables can be integrated with water quality using simple metrics, namely the lagoon quality index (LWQI), which integrates oxygen saturation, dissolved inorganic N and P, phytoplankton chlorophyll-*a*, macro-algal and sea-grass coverage (Giordani et al., 2009). LQWI is a modified version of the WQI, in which utility functions and weight criteria are used to transform measured variables into quality scores (Cude, 2001; Said et al., 2004).

Another approach is that used in ASSETS (2009)-Assessment of Estuarine Trophic Status. This builds on the U.S. National Estuarine Eutrophication Assessment (NEEA) developed by National Oceanic and Atmospheric Administration (NOAA). Although developed for estuaries, the method has also been tested in SECS, including the Ria Formosa, and several USA lagoons (Nobre et al., 2005). However, ASSETS does not include the benthos or fish that are required BQE under the WFD.

The conceptual model of eutrophication is still evolving (Cloern, 2001) and knowledge gaps can be identified, *e.g.* variations in nutrient limitation along spatial and temporal scales, and nutrient limitation as a trigger of potentially harmful or bloom-forming phytoplankton species. This is mainly because SECS respond to increased nutrient loading in different ways, depending on the intrinsic buffering mechanisms of the ecosystem. There is a general need to develop more and better quantitative relations between vegetation indicators and water quality. The current quantitative relations have only low predictive power and do not provide sufficient foundation for assessing water quality or predicting quality future.

Metal pollution and organic pollution may even impact SECS with little industrial development. A good example is the Ria Formosa (Bebianno, 1995). A wide range of contaminants was detected including metals such as cadmium, anti-foulants such as the banned tri-butyl Tin (Coelho et al., 2002), polyaromatic hydrocarbons (PAHs) (Barreira et al., 2007) and polychlorinated biphenyls (PCBs) (Barreira et al., 2005). Other lagoons with greater anthropogenic pressures include the Venice Lagoon, which receives dioxin-like compounds and metals from industrial processes. Mercury and PAHs are responsible for ecotoxic effects in the Aveiro Lagoon, Portugal. Organochlorine pesticides and PCBs are having measurable effects in the Orbetello Lagoon, Italy.

The pressures from pollutants and contaminants may be exacerbated by the residence time and poor flushing in SECS as well as effects such as bio-accumulation and bio-magnification (Carafa et al., 2007, 2009). Some compounds are persistent and subject to bio-accumulation throughout trophic compartments of the food web in coastal lagoons (Pérez-Ruzafa et al., 2000). Some of these compounds accumulate in the fatty tissues of animals thereby contaminating seafood. Compounds that decompose can produce degradation products that are known to have adverse biological effects. Eco-toxicological studies (Coelho et al., 2001; Serafim et al., 2002) highlight the **pressures** of organic compounds to the **state** of the biota in the lagoons.

5.2.2. Pressures from aquaculture

Animal densities in shellfish aquaculture maybe two orders of magnitude greater in farmed areas than in natural populations, which enhances bio-deposition rates of organic matter from 10 to 100 fold (Graf and Rosenberg, 1997). Field studies have addressed the influence of high densities of clams and mussels on different ecosystem components and processes (Mazouni et al., 1996; Bartoli et al., 2001a,b; Nizzoli et al., 2005, 2006a,b). The extent of **pressures** from shellfish farming depends upon the ecological characteristics of the different species as well as being greatly influenced by farming practices.

Bivalve filter feeders are recognised for having important functional roles in aquatic ecosystems, as they affect both the water column and bottom substrates, modifying the particulate to dissolved nutrient ratio, changing sediment composition (Prins et al., 1998; Newell, 2004) and controlling the benthic–pelagic coupling (Strayer et al., 1999). Several studies demonstrate that mollusc farming also stimulates mineralisation rates, sediment oxygen consumption, P and ammonium ($NH4^+$) recycling to the water column (Baudinet et al., 1990; Gilbert et al., 1997; Kaiser et al., 1998; Magni et al., 2000). The grazing of phytoplankton by bivalves effectively removes N from the system, a valuable ecosystem service (Ferreira et al., 2009).

Shellfish aquaculture stimulates benthic metabolism, and mussel-farming also induces anaerobic processes and sediment reduction. There are large differences between the effects of clam and mussel-farming. Clams and infaunal species stimulate the transfer of both organic matter and oxygen to the sediment, whereas suspended culture practices of mussels enhance only organic matter enrichment. Under these circumstances, a feedback loop establishes between biogeochemical processes and the farmed molluscs. Mussels are usually farmed in suspended cultures, and influence benthic fluxes only through bio-deposition, which leads to an organic matter enrichment in the upper sediment horizon. In addition, the mussel bulk on the ropes consumes oxygen and releases NH_4^+ at very high rates, directly in the water column. In contrast, the infaunal manila clam (*R. philippinarum*) is completely buried in the sediment, where it causes particle reworking and sediment mixing and supports greater suspended organic carbon, dissolved inorganic carbon and NH_4^+ fluxes than in natural benthic systems.

Mollusc harvesting can also induce perturbations in biogeochemical cycling. For example, clams and infaunal species are sometimes harvested by dredging the upper sediment horizon, which is some 10 cm thick. Dredging causes an amplification of benthic fluxes, which is transient and followed by sediment reoxidation and further stabilisation (Viaroli et al., 2003). The extent of such perturbation also depends on the dredging technique (Castadelli et al., 2003). Manual harvesting, as practiced in the Ria Formosa, seems to have only local and to some extent negligible effects, whilst mechanical dredging, as practiced in the Venice lagoon, leads to great effects on sea-floor integrity with a relevant loss of the finest sediment fraction and a strong modification of sediment texture (Pranovi et al., 2003).

5.2.3. Pressures from bio-pollution and invasive species

Aquatic invasive species (AIS) are a descriptor of the Marine Strategy Directive but this is applicable to offshore waters and does not include SECS. AIS are considered as biological pollution as a concern for the ecological **state** of SECS, but also **impact** human welfare through threats to human health and socio-economic costs (Leppäkoski, 2002; Elliott, 2003). Climate change stresses some species beyond the physiological tolerance limits and other species can expand their geographical distribution. This increases the intensity of invasion of SECS by exotic species or non-indigenous species (NIS). Some species are deliberately introduced by a sector, such as mariculture (*e.g.* Manila clam in Venice). Shipping and maritime transport is the main vector for the introduction of exotics to polluted harbours.

SECS are also gateways for bio-invasions where invading species represent an emerging threat to maintaining high biodiversity. Today the biota of SECS are exposed to each other because of the breakdown in geographical barriers due to ship traffic, leading to an exchange of species and further homogenisation of aquatic animal and plant life worldwide. SECS are interesting areas for studies on alien species since there are steep gradients in the physico-chemical environment, biological communities, degree of pollution and other human activities. SECS biota is a mixture of native and non-native species of marine, brackish and freshwater origin. In comparison with other coastal inlets, European SECS seem to be more easily invaded, and the effects of introductions are more evident (Olenin and Leppäkoski, 1999).

SECS are considered as important bridgeheads for NIS, since most harbours worldwide are located at river mouths or in coastal inlets of reduced salinity. Low native species richness and diversity of functions provide available empty niches that facilitate the successful introduction and establishment of AIS in SECS (Paavola et al., 2005). There is evidence of significant changes in the environmental and ecological status of SECS caused by alien species along the coasts of the Mediterranean, Black, North and Baltic Seas.

Some AIS can be of commercial interest and a significant number of exotic species have become valuable resources. There are 19 established alien species of marine invertebrates in the Venice Lagoon (Pranovi et al., 2006). The Manila clam (*R. philippinarum*) was introduced in 1983 and is the most successful of these species. Some non-native fish (*e.g.* round goby *Neogobius melanostomus*) are of interest to anglers, while both larval and adult stages of several AIS are important food for native commercial fish.

Research into marine invasion biology focuses on the threats AIS pose to the most infested SECS as well as ports and estuaries. Climatic changes have resulted in changes of biodiversity due to the introduction and establishment of exotic species (EEA, 2006). **Impacts** on human welfare include economic losses caused by the massive invasion of NIS species. Polluted or physically degraded environments are more prone to invasion than pristine sites, so the occurrence of NIS is an indicator of changes in environmental and ecosystem status (**state**). Therefore, the tolerance of exotic species to pollution makes them good candidates for assessing Ecological Quality Status (see Section 5.3.3 on benthos). This is a strategic issue for both the WFD and the European Thematic Strategy on the Protection and Conservation of the Marine Environment (EEA, 2006).

5.2.4. *Pressures from habitat loss*

SECS are highly productive and structurally complex ecosystems, typically with multiple habitats such as sand dunes, mudflats, creeks and channels and associated eco-tones such as salt marsh and sea-grass meadows. SECS in Europe experience pressures due to an increasing human population and a still more intensified agriculture industry. Urbanisation (**driver**) along coastlines result in direct habitat loss (**pressure**), such as the loss of salt marshes by the construction of airports, UWWT plants, salt pans, aquaculture ponds and marinas.

The shipping and maritime transport sectors (**drivers**) cause pressures on habitats because of the construction of ports and maritime structures as well as dredging, that changes both sea-floor integrity and hydrology.

Urbanisation (**driver**) along coastlines also results in the loss of wetland and sea-grass habitat by the construction of resorts and marinas. Nutrient loading (**pressure**) from agriculture and urban sewage indirectly affects habitats. Increased phytoplankton production decreases the light conditions for the benthic flora and submerged aquatic vegetation (SAV), such as sea-grasses. SECS all over Europe are therefore facing loss of habitats despite the introduction of the Habitat Directive.

5.3. *State of European SECS*

Elliott et al. (2006) have revised the DPSIR framework so that State is now interpreted as “State change”. The European WFD-Water Framework Directive uses “Status” and includes physico-chemical “elements” and biological “elements” to describe the ecological “status” of all water bodies, including coastal lagoons. These cannot be considered separately, for example changes in temperature may also affect the distribution of species (also see Section 5.2.3) and changes in salinity affect osmotic pressure and hence the survival of halo-tolerant species. The WFD “elements” are the approach used in the following sections.

Key environmental variables (*e.g.* T °C, S, pH and Eh) in SECS are all subject to global changes due to sea-surface warming, changes in hydrology and evaporation, ocean acidification and increases in hypoxic events and zones. These physico-chemical quality elements are fundamental to the chemical state of SECS and also control chemical equilibria at the sediment interface (*e.g.* phosphate and metals).

5.3.1. *Nutrient condition*

In SECS, biogeochemical processes take place mainly in the benthic compartment at the water–sediment interface, due to the low water volume to sediment surface ratio (Golterman, 1995a,b; Castel et al., 1996). Biogeochemistry and biogeochemical cycles in lagoons have been reviewed in Kjerfve (1994). Due to the shallowness and to the inherent high sediment surface area to water volume ratios, the sediment and its resident benthic communities are the most sensitive components of the ecosystems and act as triggers for processes in the water column. Sedimentary and benthic elements support also organic matter and nutrient recycling and the reactivity of biogeochemical buffers (Golterman, 1995a,b; de Wit et al., 2001).

Phosphate equilibrium at the sediment interface and P cycling depends mainly on geochemical reactions with calcium, carbonates, iron (Fe), aluminium and humic compounds (Golterman, 1995a; Jensen et al., 1998). However, only non-refractory organic P and Fe bound P are considered to contribute to P exchanges between sediment and water, through organic matter decomposition and the reduction of ferric iron species (Golterman, 2001; Rozan et al., 2002). The amorphous ferric iron species (especially FeOOH) retain phosphates in the solid phase through the formation of complexes and/or loose-binding. Under persistent anoxic conditions, bacterial sulphate reduction stimulates the iron reduction thus favouring phosphate release to pore-water and thereby to the water column (Giordani et al., 1996; Stal et al., 1996; Heijs et al., 2000). In the sulphide/iron-monosulphide/pyrite system, a series of reactions of iron with sulphides lead to the formation of the highly insoluble FeS and FeS_2, which not only represents a potential mechanism for the removal of toxic hydrogen sulphide, but also contrast the phosphate retention by sediment (Golterman, 1995a,b; Heijs and van Gemerden, 2000; de Wit et al., 2001).

The sedimentary biogeochemistry of sulphur and iron has been reviewed focussing on biological processes mediated by both bacteria and benthic fauna (Meysman and Middleburg, 2005; Rickard and Morse, 2005).

N cycling and transformations within coastal lagoons depend on the interactions between benthic vegetation and microbial processes (Eyre and Ferguson, 2002; Risgaard-Petersen, 2003) and varies between different primary producers groups (Pedersen et al., 2004; Sundbäck and MacGlathery, 2005). Nitrogen uptake and retention within biomass depends upon life cycle and tissue recalcitrance (Buchsbaum et al., 1991; Banta et al., 2004). Therefore, sea-grasses act as a nitrogen sink, keeping de-nitrification rates and benthic effluxes of nitrate and ammonium almost negligible (Risgaard-Petersen and Jensen, 1997; Welsh et al., 2000; Bartoli et al., 2001a,b; Eyre and Ferguson, 2002). Under these circumstances, SECS store and retain N. Micro-phytobenthos is thought to favour the development of sharp gradients at the water–sediment interface (Sundbäck and MacGlathery, 2005), where oxic to anoxic gradients can be established that promote coupled bacterial nitrification–de-nitrification processes. Under these conditions, N is lost from the water mass, to some extent counteracting the external loading and avoiding ecosystem deterioration. However, the stability of the MPB system depends on physical perturbation (*e.g.* turbulence and re-suspension). Nitrification–de-nitrification coupling is regulated by the autotrophy to heterotrophy ratio of the MPB system (Risgaard-Petersen, 2003).

Microbial and geochemical processes at the water–sediment interface are influenced by bio-turbation that can support primary productivity and eutrophication processes with internal loadings (see Kristensen et al., 2005 for a detailed review). Sediment porosity, vertical distribution and quality of organic matter, mineralisation rates and solute transfer across the water–sediment interface are greatly influenced by burrowing, particle reworking, as well as ventilation and irrigation activities from benthic macrofauna. Moreover, the network of burrows walls within the uppermost sediment horizons expands the sediment surface area to volume ratio, thus enhancing sediment-water exchanges.

5.3.2. *Oxygen condition*

The oxygen condition of SECS is controlled by hydrological conditions, bio-turbation by benthic infauna, geochemical reactions and biogeochemical reactions (see Kristensen et al., 2005 for review). These include primary production, organic matter decomposition, microbial mineralisation and oxidation processes.

Pressures from sewage discharges and decaying autochthonous material significantly increases the biochemical oxygen demand of SECS and leads to large oscillations in the dissolved oxygen content. During daylight hours, algal photosynthesis fuelled by nutrients and light produces considerable oxygen and saturation values in excess of 200% are possible. However, at night, respiration depletes the oxygen and anoxic conditions in the waters can occur. When this is coupled to the relatively poor flushing of the inner waters of lagoons, poor water conditions might exist. Studies in the Ria Formosa Lagoon (Portugal) have demonstrated the transport and deposition of sewage derived organic matter at sites remote from discharge pipes for UWWT (Mudge and Duce, 2005). These sites tend to be where the currents allow the settling of suspended particles. The resulting sediments in many lagoons across Europe are frequently black in colour due to the presence of metal sulphides in anoxic conditions and, in some cases, may produce hydrogen sulphide gas that is toxic to aerobes. When this occurs, there is a fundamental change in ecological status from aerobic to anaerobic metabolism.

Different plant community typology makes a substantial difference in biogeochemical cycling, primarily in oxygen and dissolved inorganic carbon production and consumption rates (Pedersen et al., 2004). The biogeochemical conditions in the water column and the sediments experience large diurnal fluctuations with oxygen depletion events during the night (Krause-Jensen et al., 1999), resulting in a poor environment for benthic flora and fauna and loss of biodiversity. Most benthic flora and fauna are sensitive to low oxygen concentrations and are lost under severe depletions as observed for sea-grasses (Carlson et al., 1994; Rask et al., 1999) and mussel beds. Sea-grasses are sensitive to oxygen depletion in the water column, as this stops the oxygen transportation through the leaves to the below-ground system, which turn anoxic (Pedersen et al., 2004). The roots can survive short periods of anoxia by shifting to anaerobic metabolism, but if there is an invasion of sulphide from the sediment at the same time, the growth is reduced and mortality increases (Goodman et al., 1995; Holmer and Bondgaard, 2001).

Eutrophication often results in organic enrichment of lagoons increasing the risk of oxygen depletion events in particular in deep and stratified lagoons (Ellegaard et al., 2006). MPB is responsible for oxygen production at the water–sediment interface, thus allowing oxygen penetration in the superficial sediment horizon, as well as oxygen release through the whole water column. Rhizophytes and benthic sessile macro-algae deliver oxygen through the water column allowing its oxygenation. Vegetation type and morphology determine a physical stratification of the water mass, *e.g.* floating foliose thalli induce water lamination and stratification, with oxygen over-saturation in the superficial water mass above thalli and anoxia in the deeper layers beneath thalli. Moreover, oxygen is released in the root system through the radial oxygen release. In healthy sea-grass meadows, production and respiration rates are usually well balanced with smoothed fluctuations, whilst in macro-algal mats a wide oscillation can occur with super-saturation followed by anoxia (Viaroli et al., 2010).

Carbon cycling is to a great extent controlled by organic matter production and microbial decomposition processes (Banta et al., 2004). Organic matter decomposition is not only regulated by quantity but also depends on quality and recalcitrance, elemental and macro-molecular composition, growth rates and life cycles of benthic vegetation (Buchsbaum et al., 1991; Enriquez et al., 1993). Sea-grasses are composed of more refractory components in relation to macro-algae, which also grow and decompose much faster, leading to shifts in oxygen metabolism (Neubauer et al., 2004; Lomstein et al., 2006).

Dystrophic crises are one of the strongest perturbations of the biogeochemical metabolism. These often occur in the summer in Mediterranean SECS, when the accumulated macro-algal biomass undergoes rapid decomposition, causing anoxia and the release of both sulphides and phosphorus to the water column (Izzo and Hull, 1991; Castel et al., 1996; Solidoro et al., 1997; Viaroli and Christian, 2003; Viaroli et al., 2010).

5.3.3. *Assessment of ecological status*

The WFD (2000/60/EC) sets the legal requirements for the classification of ecological status of SECS in its provisions for transitional waters. All European Union member states (MS), as well as some other countries such as Norway, have adopted this legislation. The WFD requires the ecological status classification to reflect changes in the structure of the biological communities and in the overall ecosystem functioning as a consequence of anthropogenic pressures (*e.g.* nutrient loading, acidification, toxic and hazardous contaminants, physical habitat alterations, etc.). Therefore, a priority is the development of systems for the monitoring, assessment and classification of ecological status to inform management strategies to ensure the sustainable use of SECS, but this has proved to be particularly difficult due to the complex dynamics of SECS.

The ecological status is defined in terms of the degree of deviation from reference conditions of a surface water body type. Thus, the process for the assessment of ecological status requires several steps including: (1) grouping of water bodies into types (for lagoons, two alternative typology systems are proposed in Annex V, Article 1.2.3); (2) determination of type-specific reference conditions for the biological quality elements; and (3) classification using Ecological Quality Ratios (EQR). The EQR is defined as the ratio between the type-specific reference value and the observed value for a given biological metric and is divided into five quality classes (high, good, moderate, poor, and bad) based on biological quality elements and to which the principle of '*one out all out*' applies.

However, the high spatio-temporal variability of SECS may be higher than that expected in more open coastal assemblages (Pérez-Ruzafa et al., 2007a). This means that the patterns in species and community distribution, and the sources of such variability, must be taken into account when designing sampling strategies to evaluate anthropogenic pressures or to establish reference conditions. This is necessary to distinguish when changes in the **state** of the ecosystem are caused by human **pressure** and not due to natural variability. The quality status of the biological elements is supported by the quality status of the hydro-morphological and physico-chemical elements (see Sections 5.3.1 and 5.3.2), therefore for SECS to classify in high or good ecological status, the quality of the supporting elements should also be high or good.

At the time of the adoption of the WFD (end of 2000), there were no comprehensive ecological classification systems for SECS and relatively little attention had been given to these ecosystems with regard to the development of ecological indicators (Kuuppo et al., 2006). However, the need for new tools for the assessment of ecological status in the WFD resulted in an impetus to develop ecological assessment methodologies for SECS (*e.g.* Bricker et al., 2003; Newton et al., 2003; de Jong, 2004; Marín-Guirao et al., 2005; Koutrakis et al., 2005; Mouillot et al., 2005; Nobre et al., 2005; Foden and Brazier, 2007; Austoni et al., 2007; Reizopoulou and Nicolaidou, 2007). Nevertheless, there are still gaps, particularly with respect to the ecological assessment of fish, and in the determination of reference conditions for all biological elements.

At first, the quality assessment systems of SECS were mostly targeted to detect changes in the trophic **state** as resulting from increased **pressures** from nutrient loading. Depending on the hydrologic and hydrodynamic characteristics, such assessments are either based on the phytoplankton assessment using simple Vollenweider type relationships between input rate of nutrients and mean Chl-*a* concentration (Vollenweider et al., 1998; Tett et al., 2003), or based on SAV, angiosperms, macro-algae and epibenthic micro-algae (Short and Wyllie-Echeverria, 1996; de Jong, 2004). Supporting physico-chemical values for oxygen condition and transparency, and zooplankton species and abundance are often taken account in a final evaluation of the trophic status (*e.g.* Margonski and Horbowa, 2003).

5.3.3.1. Phytoplankton status. Phytoplankton metrics were proposed during the EU WFD intercalibration exercise for the assessment of the phytoplankton biological quality element in coastal waters and transitional waters, which include most SECS (Loureiro et al., 2006; Goela et al., 2009).

5.3.3.2. Aquatic flora. The aquatic flora includes many different plant assemblages. These are micro-phytobenthos, macro-phyte algae, sea-grasses and salt marsh plants, all of which are important in SECS. Benthic vegetation and MPB form the basis of community structure and control ecosystem functioning (McGlatery et al., 2004; Sundbäck and MacGlathery, 2005) in SECS. The community typology depends on nutrient loading, water depth and flushing rate (Dahlgreen and Kautsky, 2004). In nutrient poor and well flushed ecosystems, rhizophytes dominate until they are limited by light penetration and turbidity (depth effect). Increasing nutrient loading **pressures** determine a shift towards the development of epiphytic filamentous macro-algae, whilst well flushed and high-load water masses become dominated by phytoplankton. The community evolution can be represented with a four-phase succession model (Schramm, 1999), where pristine and well preserved SECS are dominated by perennial sea-grasses and rhizophytes taking advantage of nutrient supply from sediment (Borum, 1996; Hemminga, 1998). An increasing nutrient input to the water column favours phytoplankton and epiphytic algae, which can damage sea-grasses until they are displaced and substituted either by floating, opportunistic macro-algae or by phytoplankton communities. In the later stages and/or in turbid waters, pico-plankton and cyanobacteria species tend to dominate.

The European coastline includes many SECS where the depth of the euphotic zone is greater than the compensation depth, or which are optically shallow and in which sea-bed primary production can be important. Due to the shallow depths, light reaches the substratum and this provides suitable conditions for the development of important benthic algal communities. MPB chlorophyll concentrations are variable throughout Europe, ranging from 60 to 125 mg chl m^{-2} at Gulf of Fos in the Mediterranean Sea (Barranguet, 1997), 80 to 500 mg chl m^{-2} in the Ria Formosa lagoon on the Atlantic coast (Brito et al., 2009a), 21 to 939 mg chl m^{-2} at the Westerschelde estuary in the North Sea (Hamels et al., 1998) and 20 to 100 mg chl m^{-2} in (Irigoien and Castel, 1997). Lucas et al. (2001) showed that the benthic contribution of MPB cells to the total diatom abundance in the water column can exceed 42% over the flood-ebb. The re-suspension phenomenon is therefore essential in the benthic–pelagic interaction. In addition, MPB cells are able to avoid photo-inhibition due to a continuous and rapid pattern of individual specimens moving upwards and downwards within the top layer of the sediment (Kromkamp et al., 1998). This can be an evolutionary advantage, leading to higher rates of biofilm productivity (Consalvey et al., 2005).

The WFD assessment of water column parameters and macro-algal biomass therefore misses key components in shallow lagoons (Brito et al., 2010). Although the importance of the micro-phytobenthos has been studied to evaluate ecosystem changes, it is not used as an indicator of ecological quality in coastal waters. Diatom metrics could be developed for SECS to provide a more comprehensive ecological quality assessment, as in the case of freshwater systems. Furthermore, the understanding of sediment-water interactions, such as the re-suspension phenomenon is desirable.

Phytoplankton blooms in the nutrient rich Baltic Sea area and in Danish SECS results in shading and the consequent decrease in most light demanding benthic flora species, specifically the sea-grasses (>11%) compared to macro-algae (<7%) and phytoplankton (<1%, Nielsen et al., 2002). Thus, macro-algae and phytoplankton outcompete sea-grasses in many SECS. The successful macro-algae are often fast growing, opportunistic species, such as the leafy *Ulva* sp. or thread-like algae (*e.g. Chaetomorpha* sp.), especially in nutrient rich lagoons. These species benefit from enhanced nutrient availability in the water column as well as a fast regeneration from the sediments (Valiela et al., 1997; Sundbäck et al., 2003). These types of macro-algae may contribute to a negative feedback loop, when the large biomass decomposes fast and the nutrients become readily available in the water column (Flindt et al., 1999; Solidoro et al., 1997). Oxygen depletion and sulphide invasion are considered as important factors for such dystrophic crises (see 5.3.2) and die-back events (Carlson et al., 1994; Rask et al., 1999; Holmer and Bondgaard, 2001). Recent compilations of the depth limits of sea-grasses show that the maximum depth of sea-grass distribution is now lower compared to earlier compilations a decade ago, indicating that the light requirements has increased. One reason may be an increased demand for oxygen and thus higher rates of photosynthesis due to sulphide stress in the sediments (Duarte et al., 2007). Several studies show loss of sea-grass habitat due to the invasion of large and fast growing macro-algae, such as *Sargassum* spp. into SECS (Thomsen et al., 2006) and this is expected to become an increasing pressure in the future. The increased use of herbicides by agriculture in the catchment is another pressure contributing to the decline of sea-grasses.

Macrophyte-based indicators have been developed including the Ecological Evaluation Index − EEI (Orfanidis et al., 2001, 2003) and the IFREMER's classification scheme (Souchu et al., 2000). The EEI uses benthic seaweeds as bio-indicators of ecosystem changes from pristine status (Ecological Status Group I, ESG I) to deteriorated status (ESG II). ESG I includes species with low growth rates and long life cycles, whereas ESG II includes opportunistic species with high growth rates, short life cycles and high biodegradability rates. The IFREMER classification scheme aims at implementing an operational tool for assessing eutrophication levels in French Mediterranean SECS. The scheme allows the classification into five quality levels, which correspond to different macro-phyte communities spanning climax species, mainly phenerogams, phytoplankton and macro-algae. The macro-phyte index is then integrated with data on zoo-benthos, sediment and water quality. A good agreement between exergy and macrophyte indicators has been demonstrated (Austoni et al. 2007), although a wealth of problems exist when considering animal species, especially vertebrates.

5.3.3.3. Status of benthos. Macro-faunal assemblage composition has been described in many Mediterranean SECS (Mars, 1966; Casabianca et al., 1973; Amanieu et al., 1977, 1981; Frisoni et al., 1983; Fresi et al., 1985; Capaccioni et al., 1987; Diviacco and Bianchi, 1987; Bianchi, 1988; Lardicci et al., 1993; Guelorget et al., 1994; Millet and Guelorget, 1994; Arvanitidis et al., 2005; Mogias and Kevrekidis, 2005; Chaouti and Bayed, 2008). Some soft-bottom benthic macro-invertebrate metrics have been developed specifically for nutrient pressure and organic pollution of SECS. They include measures of diversity (*e.g.* species−area relationships, Sabetta et al., 2007), community structure (*e.g.* index of size distribution, Reizopoulou and Nicolaidou, 2007) and composition (*e.g.* Morgana and Naviglio, 1995; Koutsoubas et al., 2000; Ponti and Abbiati, 2004; Bandelj et al., 2012). A number of indices have also been developed taking into account the benthic macro-fauna (Grall and Chauvaud, 2002; Magni et al., 2005). Among others, AMBI (Borja et al., 2003) and BENTIX (Simboura and Zenetos, 2002) indices are widely used in marine coastal areas. Both are based on the principle of the ecological identity of benthic species according to their response to pollution. Nevertheless, although they produce similar results, there are discrepancies observed in the scoring of species and further restrictions to their use in certain environments. A useful assessment is provided by the taxonomic distinctness, which has the potential to evaluate the integrity of benthic communities in relation to anthropogenic disturbances (Warwick and Clarke, 2001), although its applicability to SECS has been discussed (Salas et al., 2006). Attributes of these measures of biodiversity are sample-size independence, low sensitivity to data noise, not influenced by natural controlling factors (*e.g.* changes in salinity), and high sensitivity to detection of pollution impacts.

SECS have a great internal patchiness and heterogeneity that can either amplify or bias the stressor effects. Examples are given by both taxonomic richness and body size which are chiefly related to the surface area (Sabetta et al., 2007). Nonetheless, the size distribution within the macro-zoobenthic community seems a simple and promising tool for assessing the degree of disturbance and the ecological status of SECS (Reizopoulou and Nicolaidou, 2007).

Further indicators of either state or health of benthic communities have been implemented by the IOC-Intergovernmental Oceanographic Commission Study Group on Benthic Indicators (http://www.ioc.unesco.org/benthicindicators). These include relevant synoptic information on benthic faunal condition (*e.g.* measures of community composition), controlling natural abiotic factors (*e.g.* sediment organic matter), and levels of contaminants and these were tested worldwide (Hyland et al., 2005). This approach has also been applied to SECS using organic carbon as a tracer of stressors against the benthic community (Magni et al., 2009).

The benthos is a BQE of the WFD and has been identified as a suitable ecological group for monitoring the effects of pollution. However, most studies have utilised the macro-fauna exclusively, largely ignoring the meio-fauna. This is mainly because meio-fauna are considered to be a taxonomically difficult group. Heip et al. (1988) present some of the potential advantages of meio-fauna over macro-fauna in pollution monitoring. In particular, meio-fauna are not affected by physical disturbance to the same degree as macro-fauna. The **pressures** from discharges change the **state** of the biological communities and these changes can be used diagnostically to identify the pollutant sources. Although meio-fauna may be inherently more stable than macro-fauna, **pressure** from pollutants results in a more rapid change of **state** in the meio-fauna because of their shorter generation times. The field response of macro-benthic abundances, biomass and diversity to organic pollution is comparatively well documented (Pearson and Rosenberg, 1978; Warwick et al., 1987) but studies of the effects of pollution on meio-fauna are rarer (*e.g.* Marcotte and Coull, 1974; Lambshead, 1986; Moore and Pearson, 1986; Moore, 1987; Bodin, 1988; Warwick et al., 1988; Austen et al., 1989).

Examples of SECS that are well studied with regard to benthic NIS are in the Black Sea countries the Razim-Sinoe lagoon (RO) system (Vadineanu et al., 1997), in the Mediterranean Europe, Messolongi (GR), Venice (IT) and Thau (FR) Lagoons (Pranovi et al., 2006; Corriero et al., 2007), the Northeast Atlantic Ria Formosa (PT), and the coastal lagoons of the southeastern Baltic: Curonian (LT, RU), Vistula (RU, PL) and Szczecin (PL, DE) lagoons (Balloon, 2007) and the lagoons of the German Baltic coast (Gollasch and Nehring, 2006). Many SECS are connected to both the sea alone and rivers. There may be connections with man-made canals and with other neighbouring water bodies. Thus dispersal corridors from and to inland waters have been opened (see Section 5.2.3). In the Baltic, the Vistula and Curonian lagoons became historically important nodes for further spread of the Ponto-Caspian zebra mussel *Dreissena polymorpha*; most likely the introduction of *Dreissena* into these lagoons in the early 1800s was due to transportation of mussels attached to timber rafts. The next "destination" of the species was London (1824) and Amsterdam (1826), at that time the recipient ports for the Baltic timber trade (Olenin et al., 1999). Only after that, the species started to spread in inland waters of western and central Europe (Lithuanian Invasive Species Database, 2007).

Introduced species comprise 23% of the total (macro-algal) flora of Thau lagoon, France (Verlaque, 2001, cit. in EEA, 2006), >40% of the benthic fauna of the Curonian lagoon (Olenin and Leppäkoski, 1999), 20% of the estuarine biota in the North Sea (Reise et al., 2002) and 18% of the total biota in the eastern Bothnian Sea (Olenin and Leppäkoski, 1999). The Curonian lagoon, shared by Lithuania and Russia, is one of the well-investigated SECS in Europe. Here the Alien Invasive Species (AIS) richness (number of benthic AIS per habitat) in lagoon habitats was significantly higher than in the sea. The highest percentage of AIS biomass (relative to native species biomass) occurred in artificial hard bottom habitats (up to

97%), in the sandy littoral (94%) and in the zebra mussel shell deposits habitat in the lagoon (76%), while the highest proportion (33%) of invasive species occurred in sandy littoral and euphotic zone habitats (Zaiko et al., 2007). In the lagoon, the zebra mussel *D. polymorpha* is a dominant species (mean biomass 880 g m^{-2}, 86% of total), the mussel beds covering 23% of the lagoon bottom area (Olenin, 1997). In the Vistula lagoon, the soft-bottom community structure was totally changed by the North American polychaete *Marenzelleria* sp., when it became a biomass dominant species over all sandy and muddy habitats in mid-1990s, making up to 95% of total benthic biomass (Zmudzinski, 1996). Ecological impacts include habitat modifying ability. AIS may alter the physical and chemical structure of SECS sediments by, *e.g.* production of calcareous shells, burrowing and trapping of particles.

In the Mediterranean SECS, the Manila clam *R. philippinarum*, in addition to out-competing native species, have impacted the physical environment because their harvesting has led to increased loads of suspended material (Occhipinti Ambrogi, 2002). Changes in macro-benthic community (1968–1999) of the lagoon resulting from the introduction of *R. philippinarum* and subsequent harvesting have been assessed (Pranovi et al., 2006). A sharp reduction of all other filter feeder bivalves was documented and it was possible to estimate that the filtration capacity of the benthic community had more than doubled. This has altered the functioning of the ecosystem, resulting in a stronger benthic–pelagic coupling. In this context, *R. philippinarum* gains control of the system. As a result, the Venice lagoon ecosystem has entered into a new state, probably more resistant but less resilient, with implications for future management choices (Pranovi et al., 2006). Interestingly, there have been several unauthorised attempts to introduce *R. philippinarum* in the Ria Formosa, but these have been unsuccessful.

Secondary hard bottoms (underwater constructions) provide open habitats to alien fouling organisms everywhere (Glasby et al., 2007). In the Baltic, barnacles (*Balanus*) and mussels (*Dreissena*) increase the area and volume available for associated macro- and meio-fauna, and enhance detritus-based food chains by supplying their habitat with particulate detritus. In soft bottoms, *Marenzelleria* digs deeper than native species, thus increasing the thickness of the populated surface sediment layer and depth limit of bio-turbation. Shell deposits of *Dreissena* in the Curonian lagoon have changed former soft bottoms (sand or silt) into shell gravel, and created patches of hard substrate for sessile species on uniform soft bottoms on sites. *Mya* shells form a secondary hard substrate available for associated species in the sandy southeastern Baltic Sea coastal zone. Empty shells of the barnacle *Balanus improvisus* also serve as new microhabitats for small annelids, crustaceans and chironomids. Nektobenthic AIS (*e.g.* mysids and amphipods) swim actively and spend part of their time in the water column, and may also dwell within or on the bottom. They form dense populations in the Curonian lagoon, and serve as important links in energy transfer between pelagic and benthic subsystems (Olenin and Leppäkoski, 1999, and references therein).

Some functions of the AIS are unique and new for the ecosystem, whereas some were already present if indigenous species had performed this activity earlier. Examples of novel functions include: (1) *D. polymorpha* – filter feeding in oligohaline and freshwater parts of the Baltic Sea SECS where *Mytilus edulis* is absent; (2) *B. improvisus* – suspension filter feeding in the uppermost hydrolittoral zone; (3) *Marenzelleria* sp. – deep bio-turbation of the sediment; (4) *Rithropanopeus harrisii* and *Eriocheir sinensis* – epibenthic invertebrate predators and scavengers in the diluted parts of the inlets where native marine decapod crustaceans do not occur (Olenin and Leppäkoski, 1999).

Some of the most successful alien species tend to markedly alter the SECS habitats that they have invaded or were intentionally introduced to. Invaders in the species-poor native communities of SECS along the coast of the Baltic Sea manifest their ability to modify their novel habitats in several ways. They increase the three-dimensionality of the benthic habitats, broaden the food base of bottom and plankton eating fish, link benthic and pelagic subsystems, and create new micro-habitats for associated fauna. They increase of physical diversity of homogenous sandy and muddy silty bottoms by adding new space components into the benthic subsystem (*e.g.* shells, shell fragments and burrows), and create new microhabitats for associated fauna. They increase the functional diversity and bentho-pelagic linkages, and thus the 3-dimensionality of benthic subsystems. This broadens the food base of both benthos and plankton eating fish and hence modifies the impact of predation on native species. They retain more of the river input of particulate and dissolved nutrients in semi-enclosed coastal systems.

The seemingly "beneficial" contribution of non-native species to the structural and functional diversity must be carefully evaluated in relation to their capacity to compete for space and other resources with indigenous biota and with the negative influence on the settlement of larvae and juveniles of native species. The detrimental (often catastrophic) effects of alien organisms on different uses of coastal waters and impacts on human welfare are well documented. In the Baltic Sea and elsewhere, the majority of established AIS occur in SECS (Paavola et al., 2005). Thus, these areas should be considered continent wide as "hotspots" for the introduction of alien species and should be the focus of the corresponding monitoring programs. Surveillance of AIS should target all taxonomic groups and focus on high-risk sites (selected here from Genovesi and Shine, 2003; LIFE Focus, 2004). These include the main entry points for tourist arrivals (harbours and open moorings); entry points of natural dispersal pathways (coasts, border crossings of water systems shared with neighbouring countries); areas adjacent to facilities where alien species are kept in captivity or containment (fish farms, nurseries); areas where severe disturbance has occurred (land clearance, storm damage, etc.).

5.3.3.4. Status of fish. Fish assemblages have been well studied in SECS (Casabianca and Kiener, 1969; Herve and Brusle, 1980, 1981). They have also been related to confinement gradients (Mariani, 2001). Despite the apparent mobility and migratory displacements of many lagoon and estuarine fishes, the structure and composition of fish assemblages are highly sensitive to changes in lagoon morphology (Pérez-Ruzafa et al., 2007b; Franco et al., 2008) and to the impact of coastal works that modify the substrate characteristics (Pérez-Ruzafa et al., 2006). All the morphological supporting quality elements strongly influence the biological quality elements and characteristics of fish assemblages, including species richness and fishing yields (Sabetta et al., 2007; Pérez-Ruzafa et al., 2007b; Franco et al., 2008).

However, there are only a few attempts for developing fish ecological indicators in lagoons (*e.g.* Mathieson et al., 2000; Mouillot et al., 2005) and at present there is no experience in Europe in monitoring and classification of transitional waters based on ichtyofauna metrics (see Franco et al., 2008 for a first approach). The Ria Formosa is considered to be a "sheltered CW" rather than a "TW" because of the lack of significant freshwater inputs. So, fish are not included in the assessment of ecological status. This is unfortunate since the Ria Formosa had the densest population of *Hippocampus guttulatus* and *H. hippocampus* ever recorded, until recently. The seahorse project http://seahorse.fisheries.ubc.ca/ has documented the dramatic decline of these fish in the Ria Formosa (Caldwell and Vincent, 2012).

5.3.3.5. Ecological status and ecosystem change. SECS are highly sensitive to pressures and rapid shifts in ecosystem structure and function (Groffman et al., 2006). The multiple pressures on SECS may cause dramatic ecosystem changes and remediation responses are often not effective. Recent relaxations of pressures have failed to show recovery to their origins, leaving a major challenge for scientist and coastal zone managers. Several European lagoon ecosystems have entered into a new state, probably more resistant but less resilient, with implications for future management choices (*c.f.* Pranovi et al., 2006). It will be difficult or even impossible to return lagoons to their original conditions, as the impacted ecosystems have changed considerably. Danish lagoons (fjords) are examples of systems where sea-grasses are disappearing. Sediments that are organic-rich because of phytoplankton sedimentation and are also more fine-grained, may be inappropriate for seeds of sea-grasses. As the phytoplankton production increases, sea-grasses are lost from lagoons and an invasion of suspension feeders may occur, such as that observed for *M. edulis* in Danish SECS (Vinther et al., 2008). In the Ria Formosa, the sea-grasses are disappearing while opportunistic algae are proliferating. *M. edulis* not only compete with other benthic fauna and flora for substrate, but also filter feed and improve the light conditions, thus promoting benthic flora. However, the effects of mussel invasion are primarily negative in eutrophied areas with organic enriched sediments. The faeces enrich the sediments enhancing the sulphate reduction rates and thereby the sulphide stress on the sea-grasses (Vinther et al., 2008).

Another example of the consequences of agricultural **drivers** in the watershed is the regime change in the Mar Menor (S Spain). Agricultural drivers include a change in practices evolved from extensive dry crop farming to intensively irrigated crops, with increasing **pressure** of nutrient and pollutants to the lagoon. Top-down control of the trophic web exerted by ichthyo-plankton and jellyfishes (Pérez-Ruzafa et al., 2002, 2005) crop the phytoplankton and keep the water clear.

5.4. *Impacts on human welfare*

The present interpretation of DPSIR (Elliott et al., 2006) concentrates on **impact** on the human system and human welfare. It is therefore important to understand the linkages between changes in environmental and ecological **state** leading to the loss of ecosystem services and how these are linked to social and economic impacts on human welfare. The most impacted sectors are fisheries, aquaculture and tourism. For example, jellyfish affect the recreational uses of the Mar Menor for tourism leisure activities, and especially bathing. Massive blooms of jellyfish are "dead-end" species with no commercial value that negatively affect the fisheries and aquaculture in Danish lagoons (Møller and Riisgard, 2007). Magnan et al. (2009) describe the "chain of impacts" on Mediterranean SECS.

5.4.1. *Overexploitation of natural resources*

SECS in Europe have long been exploited for products to support human consumption, either through fisheries or aquaculture. Unfortunately, there are many examples of overexploitation, as fisheries regulations are some of the most difficult to enforce, even locally. Many natural resources have been lost, and coastal fisheries have declined adjacent to SECS (Myers and Worm, 2003). Products such as algae, farmed fish and shellfish have been exploited, and while the capture fisheries declined, shellfish landings increased. Capture fisheries are now a subsidiary activity in SECS.

Shellfish production has also declined due to both known and unknown reasons (Herlyn and Millat, 2000). The settling of mussels has for instance been reduced for several years in Danish lagoons (Sand-Kristensen, pers. comm.) and clam production has declined in the Ria Formosa. Overall, the sustainability of shellfish farming is not a simple function of a given environmental or economical variable, but rather results from a wide array of parameters and variables that can be evaluated only with models that integrate hydrodynamics, ecological components and economic issues (Solidoro et al., 2000; Pastres et al., 2001; Solidoro et al., 2003; Cellina et al., 2003; Zaldivar et al., 2003). For example, the sustainability of clam farming in the Sacca di Goro lagoon has been assessed by modelling exploited areas, biomass bulk, risk of anoxia and sensitivity of clam farming to *Ulva* m^{-2}, with food supply being the main limiting factor. In sheltered areas, an increase in clam production over 2 kg m^{-2} y^{-1} is not beneficial, since the gain is equivalent to or less than the cost of measures taken to control nuisance macro-algal blooms. In other words, the aquaculture productivity cannot be maximised but rather can be only optimised, taking into account a suite of environmental variables that identifies shellfish farming suitability (Vincenzi et al., 2006a,b). Yet, the suitability is assessed with functions that evaluate sediment quality, food and oxygen availability, salinity range and hydrodynamics. Blooms and anoxia may result (Marinov et al., 2007; Viaroli et al., 2008). Either a sustained standing crop or production per surface unit do not have fixed thresholds, but are function of a number of interlinked variables. For clams in open bays or highly dynamic lagoons, a sustainable threshold is close to 4 kg m^{-2}.

In the Sacca di Goro the aquaculture yield is increasing due to the Manila clam farming, whilst the indigenous *R. decussatus* has almost disappeared. In the Lagoon of Venice, the poor control is exerted over the catches of *R. philippinarum*. The introduction of this species resulted in the disappearance of the indigenous *R. decussatus* and contributed to change the **state** of the benthic community as well as significantly increasing the erosion rate in many lagoon areas.

5.5. *Societal responses, EU policies and management of SECS*

European SECS are subject to various kinds of anthropic **pressures**, which are often sources of conflicts among the different users. Balancing socio-economic interests with environmental safeguards is an extremely difficult task. In this respect, integrated management, together with the development of interdisciplinary and multi-criteria approaches, is one possible key to the sustainable, equitable and efficient development of SECS resources.

SECS are dynamic environments and human inhabitants can be threatened by floods and storm surges. As a consequence, coastal defense structures result in extensive morphological modifications, a technological **response** to control dynamics that affects the perimeter, depth and communications with the adjacent coastal waters.

Another technological response, this time to water quality, has been the construction of sewage treatment plants. Municipal sewer systems have reduced health risks and UWWT plants have significantly reduced organic matter. However, N and P diffuse inputs remain a problem, as does eutrophication.

One of the societal responses to problems in European SECS has been the implementation of EU policies and Directives by member states. The societal response to the pressures of nutrient loading and negative impacts of eutrophication in European SECS has been the introduction of legislation and management measures to reduce the nutrient availability in transitional and coastal waters. The most important Directive in this context is the WFD.

The WFD aims at ensuring the protection of European surface and ground waters, including transitional and coastal waters, from a wide range of pollution sources (point and diffused sources, among others: waste water treatment, intensive agriculture practices, industrial emissions etc.), as well as requiring Member States (M.S.) to identify vulnerable or protected areas, undertake monitoring programs and, where necessary develop remedial measures for water bodies at risk.

Monitoring of key elements of SECS and research activities studying ecological interactions are needed to continuously improve knowledge, *e.g.* the threshold concept, to optimise the management of SECS. European coastal SECS are areas of special interest to local and regional authorities that have monitored these systems with the aim of understanding and protecting their biodiversity and economic sustainability. Under the WFD, the responsibility for monitoring passes to the M.S. "Competent Authority" identified under Article 3(2) or 3(3). There are 3 levels of monitoring: surveillance monitoring; operational monitoring, and; investigative monitoring (WFD Annex V, 1.3). Which level of monitoring is used depends on the status of the water body and whether achieving "good" status is realistic.

EU policies are interlinked and can no longer be implemented alone, so each policy decision needs to be considered in the light of impacts from other sectors and policies relating to industry, transport, agriculture, energy, regional planning etc. For example, regulation of detergents and contaminants during the last three decades, has reduced the impact of urban waste on aquatic ecosystems (Howarth and Marino, 2006). There are numerous EU policies relevant to SECS: the Marine Strategy Framework Directive, the recommendations on Integrated Coastal Zone Management, the Habitat Directive 2 and the Water Framework Directive, WFD (2000/60/EC) (EU, 2000), the Urban Waste Water Treatment Directive (91/271/EEC) (EU, 1991a), Nitrates Directive (91/676/EEC) (EU, 1991b), Bathing Water Directive (2006/7/EC) (EU, 2006), Habitats Directive (92/43/EEC) (EU, 1992) and the Birds Directive (79/409/EEC) (EU, 1979). Areas designated under these Directives will have the status of Protected Areas under the WFD (Annex IV).

Furthermore, there are a number of International and Regional Conventions (OSPAR, 2001; HELCOM, Barcelona and Bucharest Conventions) addressing measures to prevent and eliminate pollution in regional Seas (Northeast Atlantic, Baltic, Mediterranean, Black Sea), as well as Conventions for major river basins such as the Rhine and Danube.

A number of Directives and regional sea conventions individually consider eutrophication in slightly different ways (either directly or indirectly) depending upon the objective of the policy, the pollution sources and the receiving waters (river, lake, groundwater, transitional and coastal waters). These require a common understanding and framework for eutrophication assessments.

New methodologies in terms of economic assessment, social requirements and resource management will be needed to optimise EU policy decisions so as to reinforce measures to protect the environment and ensure that the all environmental Directives are implemented in such a way that they have a real chance to succeed in their objectives. Modern concepts of the "system approach" to ecosystem management (Newton, 2012) include concepts of public information, public participation in decision making, social and economic considerations (Viaroli et al., 2012). Most SECS are situated in highly populated areas and are affected by human activities. Prohibitions and the transformation of SECS into reserves protected from human influence are insufficient management **responses** to ensure sustainability. Effective management requires a combination of regulation and the public understanding that SECS are natural resources that should be conserved for the benefit of present and future generations. The public must be allowed access to the SECS to explore and enjoy these systems. Management must interact with the stakeholders and users of the SECS in order to obtain the acceptance of the necessary regulations in both the lagoon and in the catchment. Decisions need to be taken in the light of not only environmental considerations, but also their economic, social, and political impacts and require the active participation of stakeholders in the decision making process. The dissemination of knowledge about SECS as important natural resources and their relation to economic activities is essential for the public to understand the basis of informed decisions. Interactions of the lagoon, the catchment and the adjacent coast must be understood for integrated management to consider the most important direct and indirect **pressures**. For integrated adaptive management using a system approach to be successful, societal **responses** should address all the **drivers**, not only some of them. For example, it may be futile to build more advanced UWWT plants if the agricultural practices in the watershed are responsible for the major **pressures** relevant to the eutrophication of transitional and coastal waters.

Control of eutrophication (and contaminants) of SECS remains one of the greatest challenges for managers and policy-makers (Schindler, 2006). The management of eutrophication in Ringkøbing Fjord, a coastal lagoon at the west coast of the mainland of Denmark (Petersen et al., 2008) is a particularly well documented example of societal responses to changes in state and resulting impacts on human welfare. Historical records go back to the late 1800s. The main driver of change was agriculture, with agricultural sources responsible for about 80% of the total nitrogen load. Large areas of land were reclaimed and watercourses were straightened to increase agricultural production in the catchment of Ringkøbing Fjord. The lower 20 km of Skjern Å, the river with the largest water flow in Denmark, was channelled in the 1960s, and wetland in the catchment was drained and decreased to about a quarter of the former area. Up to the 1970s, Ringkøbing Fjord seemed to be a healthy coastal ecosystem with relatively transparent water and a good coverage of sea-grasses submerged aquatic vegetation to a depth of 3 m. However, a change in **state** was observed from 1979 to 1981 when the eutrophication threshold of the system the lagoon was reached with increasing amounts of fast growing macro-algae and epiphytes. The lagoon ecosystem collapsed when huge amounts of phytoplankton and epiphytes shaded the bottom plants.

The **response** was both the regulation of detergents and the construction of sewage treatment plants in urban areas in the catchment that reduced the **pressure** of P and other nutrients to the lagoon in the late 1980s. Since the mid 1980s, the Danish Parliament adopted a number of action plans and strategies to reduce the nitrogen load to aquatic systems from agricultural sources. From the beginning of the 1990s this resulted in a slow but steady reduction in the total N load to Danish coastal waters (Ærtebjerg et al., 2003; Carstensen et al., 2006; Conley et al., 2007). The point source share of the total P load decreased from 60–70% in the 1980s and early 1990s to about 40%. The N load to Ringkøbing Fjord decreased 30% less than the maximum load in the beginning of the 1990s when the inter-annual variations in freshwater discharge were accounted for. However, due to a general wetter climate, a possible effect of global climate changes, the actual reduction in the nitrogen load to the lagoon is only about 15%.

The ecosystem subsequently started to recover slowly, however, the system was unstable and experienced several setbacks and the coverage of SAVs was limited to a depth of less than 1 m, despite a 50% reduction in the total P load. Meanwhile, national priorities started to change and the lower part of Skjern Å has been restored involving the re-establishment of 2.200 ha of wetlands from former agricultural land.

5.5.1. Modelling of SECS

Advances have also been made in ecological modelling including biogeochemical and ecological processes. The need to develop models for ecological and environmental management problems has been steadily increasing since the late 1970s. Indeed models provide both a keener understanding of causal relationships driving ecological functioning, and the quantitative knowledge which is required for evaluation, at ecological and economic levels, of consequences of the implementation of possible alternative scenarios of policy options. European SECS have been frequently considered in model applications, since they are valuable areas in which contrasting interests coexist and have to be balanced. SECS also exhibit complex patterns and high space and temporal variability of most environmental variables, because of the superposition of different anthropogenic and natural sources of variability.

Many mathematical models have been developed with the main aim of gaining insights into given biogeochemical and ecological processes, regardless of the relevance of the models for management purposes. In most cases, spatial variability has not been taken into consideration, but there have been attempts at coupling the processes with the transport. Nevertheless, a proper understanding of the temporal and spatial variability of hydrodynamics is fundamental to model SECS functions (Guelorget and Perthuisot, 1992; Cucco et al., 2009; Melaku Canu et al., 2012), and for any management intervention. In fact, the inclusion of a transport module is very often crucial when the models are applied to specific management problems, or when spatial variability is important in the water bodies taken into consideration, as in the majority of large SECS.

The modelling efforts first focused on eutrophication or other biogeochemical aspects, and were quite successful in the investigation of specific processes. These include: the kinetics of plankton growth (Nyholm, 1978); macro-algal proliferation and nutrients assimilation (Solidoro et al., 1997); the modality of inter and intraspecific competition and the interaction among the cycles of Nitrogen and Phosphorus and Dissolved Oxygen (Solidoro et al., 1997; Chapelle et al., 2000); and the importance of the remineralisation process (Chapelle, 1995). Other processes that have been taken into consideration include population dynamics, growth and energetic requirements of fishes, crabs, and filter feeders, with particular emphasis on marketable species frequently reared in lagoon waters, such as clams (Solidoro et al., 2000, 2003; Marinov et al., 2007), oysters (Gangnery et al., 2001; Gangnery et al., 2004a) or mussels (Gangnery et al., 2004b). Spatially explicit models are particularly useful in the analysis of the influence of the physical forcing functions and the transport process on the dynamics of biogeochemical properties (Chapelle et al., 2000; Melaku Canu et al., 2003; Plus et al., 2003; Trancoso et al., 2005; Solidoro et al., 2005b). Notable research effort has been focused on selected SECS, because of their ecological and socio-economic relevance. These are the Venice lagoon (Pastres and Solidoro, 2012), the Etang de Thau (Plus et al., 2006), the Ria Formosa (Nobre et al., 2005) and the Sacca di Goro (Zaldivar et al., 2003). The attention of the scientific community has focused on the modelling of the structure of the trophic network (Carrer and Opitz, 1999; Olsen et al., 2001; Zaldivar et al., 2003) and a more comprehensive understanding of interactions among biogeochemical cycles, primary productivity, and dynamics of higher trophic levels (Libralato and Solidoro, 2009). The results are a variety of approaches to incorporate more ecological details, for instance by releasing stoichiometric constraints and resolving community structure by the explicit representation of several classes of plankton (Baretta et al., 1995; Petihakis et al., 1999). More flexibility is made possible by allowing the model to change its structure in time in agreement to environmental conditions (Coffaro et al., 1997). More reliability is possible by including processes that were poorly quantified in the past, such as benthic–pelagic coupling (Serpa et al., 2007; Brito et al., 2009b), or by incorporating novel numerical techniques based on machine learning, artificial neural network and other data based approaches (Bandelj et al., 2009). Indices as exergy and specific exergy cope with these goals allowing the assessment of ecosystem health and ecological status (Zaldivar et al., 2010).

There are also attempts at developing bio-accumulation and ecological risk analysis models, usually by coupling of food web, physico-chemical partition, and bio-accumulation sub-models. Applications of this modelling approach (Carrer et al., 2000; Dalla Valle et al., 2004; Micheletti et al., 2008) were successful in describing the general trends, and are now being applied also to spatially explicit context (Carrer et al., 2005; Carafa et al., 2006). However, the results obtained cannot be considered as entirely satisfactory from a quantitative point of view, in particular for heavy metals and some organics. This is certainly due also to the inherent complexity of these processes, and to the scarcity of field data. Nevertheless, significant challenges still exist even when considering the more traditional aspects mentioned above reasons include: (1) the subjectivity in the definition of the functional groups of the food web (Anderson, 2005); (2) the lack of quantitative knowledge on many processes (Flynn, 2005); (3) the limited understanding of how model ecosystem structures optimised to present conditions might be adapted to future conditions.

More than 40 years after early attempts at modelling lagoons (Jørgensen, 1976; Di Toro et al., 1977), there are now several examples of management oriented studies in which reliable models are used for analysing alternative policies connected with specific ecosystem management issues. So far, most of these applications are related to eutrophication, fisheries and aquaculture. In particular, models exist of the effect of nutrient loads from watershed on trophic condition (Zaldivar et al., 2003; Plus et al., 2006) and water transparency (Jacobsen et al., 2006), in the determination of the maximum permissible loads compatible with predefined water quality target (Pastres et al., 2001). These analyses also provide interesting examples of how model results can be combined with Geographic Information Systems to explore the economic consequences of the implementation of different management **responses**. Examples of models application also include modelling-based approaches to the rational management of aquaculture activities, with particular attention to clams (Pastres et al., 2001; Melaku Canu et al., 2012), mussels (Grant et al., 2007) and oyster (Gangnery et al., 2004a) farming, and studies which download effects of global change to regional and local scale (Cossarini et al., 2008). Another important application is the use of models for assessment of systems state and inter-site comparison. Such models provide information that: (1) may not be measurable, for example indices of ecosystem functioning (Brando et al., 2004); or (2) that are not measured, for example fluxes among compartments (Solidoro et al., 2005a); or (3) the spatial distribution of some variables (Cossarini et al., 2009). In turn, this information can be used – often in combination with experimental observations – to drive other models, such as habitat suitability models (Vincenzi et al., 2006a,b), or simple screening model for rapid assessment of system **state**, as in the NEEA (Bricker et al., 1999), OSPAR (OSPAR, 2001) or ASSETS (Nobre et al., 2005) schemes.

Numerical models now constitute a valuable framework for integration and synthesis of existing knowledge about European coastal lagoons, and offer important contributions for understanding the scale of human disturbance and the potential effectiveness of restoration action.

5.5.2. Decision support system for the management of SECS

Multiple objectives can complicate the task of decision making, especially when the objectives conflict. The concept of a decision support system (DSS) is in fact extremely broad and there exists no unique definition of it. Power (1997) describes a DSS as an interactive information system intended to assist decision making activities by helping managers retrieve, summarise and analyse decision relevant data, thus improving and speeding-up the processes by which people make and communicate decisions. A DSS is both a process and a tool for solving problems that are too complex for humans alone, but usually too qualitative for only computers. Decision Support Systems cover a wide variety of systems, tools and technologies for informing and supporting decision makers.

As a process, a DSS is a systematic method of leading decision makers and other stakeholders through the task of considering all objectives and then evaluating options to identify a solution that best solves an explicit problem while satisfying the objectives to as high a degree as possible. As a tool, a DSS includes functionalities for the design of alternatives, and mechanisms for their comparative analysis, ranking, and selection on the basis of the criteria, objectives, and constraints provided by the users. A participatory approach, involving users, planners and policy-makers at all levels, is often quoted as a key factor in DSS success. Users are often unable to specify all their expectations and requirements at an early development stage, and a continuous involvement allows them to evaluate the system and contribute to its improvement.

Many decision support systems have been developed to face the problems of water-resource management. However, many of these are relatively simple information and model systems that focus on problem representation and, in most cases, "what-if" type scenario analysis.

In the framework of the EU funded project DITTY, a Decision Support Systems was developed for the management of coastal lagoons (Mocenni et al., 2009). The DITTY DSS represents a unitary framework for many DSS already designed and provides an answer to "*still open methodological questions about the development and structure of operational decision support systems with and for European decision makers in the field of water resource management*" (Mysiak et al., 2002). Several important decision support systems have been developed for specific application,s, such as Bayesian Networks based DSS for the management of natural resources (Bromley et al., 2005), MULINO-DSS for the computer-aided water-resource management (Mysiak et al., 2002, 2005), DSS for coastal areas sustainable development (Carvalho, 2002), WaterStrategyMan (WSM) DSS (ProGEA, 2013), DSS for Water-Resources Management under uncertainty (Pallottino et al., 2002) fit the general structure proposed in the DITTY DSS (see also Casini et al., 2005; Casini et al., 2007). The DSS was tested in the following lagoons: Ria Formosa (Portugal), Mar Menor (Spain), Etang de Thau (France), Sacca di Goro (Italy), and Gera (Greece). An application of the DITTY DSS to the problem of microbial contamination in the Etang de Thau is described in Loubersac et al. (2007).

6. Conclusions and the vulnerability of SECS in the context of global change

The main conclusions with respect to the vulnerability of SECS in the context of global change are outlined in this section.

6.1. SECS are sentinel systems and hotspots of coastal vulnerability at a global scale (Eisenreich, 2005; IPCC, 2007; Newton, 2012) especially vulnerable to large-scale impacts of climate change and sea-level rise

SECS exist in many areas of the world, Kjerfve 1994, Barnes 1980, and share many of the problems and issues in European SECS. SECS all over Europe have been the focus of intense research activity and legislation in the past 20 years. Nevertheless, as a consequence of their situation between land and open sea, the integrated Europe-wide WFD is failing to address European transitional and coastal waters in a unified manner. The main goal of the WFD for transitional and coastal waters as well as for all surface waters would be to achieve good ecological status. However, the environmental and ecological state of SECS is affected by human activities. These ecosystems of considerable ecological and economical value (tourism, fish farming and aquaculture etc.) are susceptible to human activities in their watersheds. Nutrient and contaminants fluxes lead to eutrophication and chemical contamination and the problems are exacerbated due to their geomorphology that reduces the exchanges with the open sea. An increase of 2–3 °C could result in the loss of 50% of SECS in the Mediterranean due to sea-level rise and storm surges (EEA, 2006). A 34 cm increase in sea level would result in the loss of about 30% of SECS globally (IPCC, 2007), and millions of coastal inhabitants would be threatened by flooding.

6.2. There is confusion about nomenclature and this hampers knowledge

The terminology used for SECS varies in different parts of the Europe and is extremely varied throughout the world. This hampers the transfer of knowledge. Seeking information in the international literature about SECS is problematic. For instance, there is a great diversity of terms used for coastal lagoons in the Americas, although only 3 languages are used (English, Spanish and Portuguese). In French the term is "bassin", as in Arcachon or "etang" as in Etang de Thau. Italian uses Laguna as in Laguna di Venezia but also other terms such as Sacca, *e.g.* Sacca di Goro. In Portuguese the term is "lagoa" for some systems and "Ria" as in Ria de Aveiro and Ria Formosa, which are not the same as the Galician rias. In Spain, the term "marismas" is used, but also "laguna", "albufera" and "mar", as in Mar Menor. Just as the north coast of the Mediterranean has many lagoon systems, the south coast from Morocco to Egypt also has many lagoons with very similar characteristics. The Indian coast has extensive lagoons on the Arabian Sea, *e.g.* the "lakes" of Kerela, and also on the Bay of Bengal, such as Lake Pulicat. The extensive lagoon systems on the east coast of the USA are known as "bays", as in the case of the Maryland bays, or "sound" as in the case of Pamlico sound. In California, both "bay" and "lagoon" are used, *e.g.* San Diego Bay and Newport Bay, but San Elijo lagoon. The term "laguna" is used in Texas and Mexico *e.g.* Laguna Madre and Laguna Pueblo Viejo. But the term varies in Latin American countries. "cienaga" is used in Columbia and Venezuela, *e.g.* Cienaga Grande de Santa Marta, but also "laguna", *e.g.* laguna de Unare, and also "bahia", as in Bahia Hondita. In Brazil, the commonly used term is "Lagoa", as in Lagoa dos Patos. In Argentina, the term is "caleta", as in Caleta Valdez.

6.3. Knowledge is important because the systems are ecologically valuable, and therefore provide important ecosystem services that support economic activities and societies

While human activities change the state of SECS, the degradation of the lagoons also impacts human welfare. SECS are valuable systems providing humans with ecosystem goods and services and sustaining European livelihoods. The range of ecosystem services provided by SECS is extensive and includes provisioning services, regulating services, supporting services. However, these may not be recognised or valued by the local inhabitants and decision makers, so that many of these are compromised by land-use changes. Despite their resilience, there is a threshold beyond which there is a change in the state of the SECS, and sometimes a regime shift that is very difficult and costly to reverse.

Key thresholds include natural resources, *e.g.* collapse of seagrass meadows or shellfish; social thresholds, *e.g.* closure of a school; infrastructure thresholds, *e.g.* loss of a bridge or a sea wall;

pollution and contamination thresholds, *e.g.* loss of contaminated seafood resources or forced evacuation of residents; unacceptable risk threshold, *e.g.* repeated heavy loss of housing, unable to afford or obtain insurance and, in extreme cases, death tolls.

Future technological innovations may mean that the ecosystem services of SECS may change to accommodate energy generation (tidal, wave, microbiofuels) and desalination.

6.4. The SECS, including their human populations, are also vulnerable to natural change including climate change and sea-level rise

The most evident effects are environmental variations in sea-level rise, sea current circulation, freshwater supplies and lagoon salinity. Most European SECS are infilling environments. However, the morphological and sedimentological evolution and stability of SECS is not well understood, and the main driving processes such as climate, relative sea-level, sediment availability and tidal action urgently need to be described and interpreted. The impact on SECS of changing relative sea-level is based only on scattered information of sediment accumulation rates. Although the INDIA (Inlet Dynamics) project yielded interesting results (Williams et al., 2003), further studies are necessary to provide a breakthrough in the understanding of coastal lagoon sedimentation. These should provide a strong basis for the evaluation of the morphological response of SECS to changing sea levels.

The sea level has risen in the order of 120 m (Newton and Icely, 2008) during the Holocene as a consequence of the melting of the large ice-caps which were formed during the preceding Weichselian ice-age. All SECS possess some degree of ability to adjust to changes in sea level. The sea level initially rose rapidly followed by a period of a decreasing rate of sea-level rise. Eustatic sea-level changes (changes associated with changes in the volume of oceanic water) are generally believed to be less than 1 m for the last three thousand years (IPCC, 2007). The rising sea level has had a strong impact on the coastal landscape and the coastlines generally moved landwards. This included a gradual translocation of barrier islands and coastlines landwards. However, thick deposits of fine-grained lagoonal deposits can be found in the SECS of the Wadden Sea area, including both tidal flat and salt marshes sequences as well as peat deposits (Hoselmann and Streif, 2004; Streif, 2004). This shows that some of the SECS had been situated roughly at the present locations in spite of the rising sea level. Consequently, in some areas the response to the rising sea level was deposition of sediments that enabled the SECS to maintain a roughly stable location with limited changes to the overall morphology of the coastal landscape.

The 4th IPCC assessment report (IPCC, 2007) estimates that the sea level by 2100 will have risen between 0.1 and 0.9 m compared to 1990, depending on the emission scenarios and numerical models which are used. The typical estimated rise is about 0.4 m by 2100. This will increase the water depth of SECS, but estimates of up to 5 m are increasingly realistic. The area of tidal flats and salt marshes will decrease in size if sedimentation is not able to keep pace with the rising sea level. The vulnerability of SECS to sea-level rise is obviously very dependent on sediment availability and local subsidence, *i.e.* SECS situated in areas with ample sediment supply will be less threatened by a sea-level rise than sites with reduced sediment supply and/or local subsidence *e.g.* Venice lagoon (Day et al., 1999; Madsen et al., 2007).

SECS usually contain large areas of tidal flats and fringing salt marshes situated within the tidal frame of the sedimentological units. Biological communities, which inhabit the sediments in these environments, are likely to be sensitive to changes in sea level. Global mean sea level has been rising at an average rate of 1–2 mm y^{-1} over the past 100 years, which is significantly larger than the rate averaged over the last several thousand years. A rise of a few mm per year by the sea is extremely important because low-lying SECS may be damaged or destroyed. The effects of sea-level rise may be aggravated if associated with other processes.

6.5. Long-term data sets are necessary for observing the changes in SECS in the context of climate change

SECS are sentinel systems that are affected by change across geographical gradients, ecosystem function, and particular stressors. There is a need for information and knowledge to support decision making for the management of SECS at the spatial local-regional scale and the decadal-temporal scale. In many cases, time series of climate records are too short and not sufficiently reliable, making it hard to formulate future scenarios for SECS. Observations of alterations on long time scale of erosion, biodiversity changes, modification in ecological processes and hydrological regimes, land use and consequent economic modification are necessary to understand changes in SECS, and these data are often not available. However, data going back centuries is available for the lagoon of Venice that show that regional land subsidence is occurring at the same time as sea-level rise. This demonstrates the importance of long data sets.

Results obtained in a downscaling experiment performed for the Lagoon of Venice (Salon et al., 2008) confirmed that changes in timing and volume of freshwater discharge and nutrient input induced by climate change can be critical in defining seasonal dynamic of biogeochemical properties. In particular, future climate projections for this specific area suggest that annual mean rain will not change much in the watershed of the lagoon of Venice, whereas the seasonal patterns will likely change, with summer and spring becoming drier and winter and autumn more rainy (Salon et al., 2008). This will potentially increase winter nutrient concentrations but -because of unfavourable timing – annual primary and secondary productions of the Lagoon of Venice will decrease and nutrient surplus will be exported to the Adriatic Sea (Cossarini et al., 2009). This impacts on clam species suitability, and potentially impacts the aquaculture of *R. philippinarum* (Melaku Canu et al., 2010).

6.6. Knowledge is needed for action and management to mitigate and adapt to detrimental environmental change (freshwater resources, ecosystem services, carbon budgets)

Climate change can induce changes in the tropho-dynamics of SECS by altering rain regime and, as a consequence, riverine inputs and run-off from drainage basin (Scavia et al., 2002). Water use around SECS, particularly in summer months in S. Europe when the population increases because of tourists, is often greater than the recharge of the aquifers. Subsidence is partially due to the over extraction of water and also leads to salt intrusion in the aquifers. The impacts on human welfare include changes in agriculture, fisheries, aquaculture and tourism. Responses must address social behaviour, lifestyles and the related anthropogenic pressures and drivers of the economy. Multidisciplinary approaches, such as ecohydrology, are important in integrating surface and groundwater management.

Ecosystem change has been observed in many SECS, especially with respect to the loss of sea-grass meadows. This has important consequences, not only for ecosystem services (such as the stabilising of sediment, the provision of nursery services for juveniles of commercial species of fish, and the production of oxygen), but also for the carbon budgets as sea-grasses sequester carbon. This signifies that SECS could turn from being carbon sinks to carbon sources.

6.7. Improvements are needed in forecasting extreme hazardous events (storm surges and floods)

The most important long-term challenge to coastal SECS in Europe will be the ongoing global and climate change. Global and local climate changes are a complex mosaic of events, acting at different time scales and affecting almost all the ecological processes (sea-level rise, water and air temperature, wind, storminess, etc.), influencing SECS and transitional environments with direct and indirect interference. Climate change scenarios predict an increased risk of extreme weather events. Ongoing sea-level rise as well as changes in precipitation in the catchment, with subsequent changes in river discharge, will increase the flooding risk in the river basin and in the SECS. An increased risk of storms and storm surges will have immediate negative effects on lagoon erosion, protection measures and tourism infrastructure (sport boat harbours, beaches, piers, promenades). Imminent and evident variations playing at small spatial and time scale in the lagoon ecosystem regard changes in flood intensity and frequency. In the Venice Lagoon, average yearly occurrence of tide peaks over 80 cm has gone from 10 cases in the first half of the century, to 40 in the second half, reaching almost 60 cases in 2000–2005. The "*Aqua Alta*" floods in December 2008 and also in 2012 illustrated the urgency of the MOSES barrier for the protection of Venice.

6.8. Technological innovation and eco-innovations are needed

Innovation in forecasting flood events, technological innovation and eco-innovation is also needed to improve the prognostic for SECS. Examples of new technologies include synthetic biology applications to architecture (Armstrong and Spiller, 2010) Examples of eco-innovation include the use of bivalves in aquaculture for eutrophication abatement (Ferreira et al., 2009).

6.9. A system approach to adaptive management is an important in management and governance response

Realistic management strategies should consider the consequences of climate and global change in the implementation of local solutions to come up with new answers to old problems. The scale of SECS and their catchment make these systems ideal for the adoption of local and regional strategies including stakeholders and participatory decision making where science can inform policy and decision makers in a nested governance framework. Because of the geographical scale of SECS, these are often managed at the local level with limited understanding of global issues and research that are relevant.

An ecosystem based, system approach to management is key to solving management challenges for SECS. In order to co-design solutions to the problems and issues for the management of SECS, it is necessary to bring together scientists from different disciplinary backgrounds and with experience at different sites, with local actors, stakeholders and decision makers. The System Approach Framework (SAF) is such a procedure that has been tested in several SECS (Newton, 2012), including the Mar Piccolo, the Lagoon of Venice, the Oder-Odra and the Etang de Thau. The DPSIR framework is also useful in a balanced analysis of SECS as complex social-ecological and economic systems. Spatial planning and an integrated approach are key tools for the management of coastal lagoons and balancing the conflicting interests and uses. Engaging public participation, for example in participatory monitoring or beach cleanup activities, provides an arena for members of the public to interact with researchers and managers and feel that they are part of the solution rather than the problem and enable behavioural changes. The role of education and dissemination is vital to establish effective science communication. This may result in zoning and set back lines to protect the lives and properties of inhabitants living in high-risk zones and vulnerable to the threat of multiple natural hazards.

An international, interdisciplinary effort can generate the results necessary for scientific advancement to achieve a global understanding of the effect of global change on SECS, and generate knowledge for large-scale policy making and the management of SECS worldwide.

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