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Medium- and Long-term Recovery of Estuarine and Coastal Ecosystems: Patterns, Rates and Restoration Effectiveness

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Abstract Many estuarine and coastal marine ecosystems have increasingly experienced degradation caused by multiple stressors. Anthropogenic pressures alter natural ecosystems and the ecosystems are not considered to have recovered unless secondary succession has returned the ecosystem to the pre-existing condition or state. However, depending upon the scales of time, space and intensity of anthropogenic disturbance, return along the historic trajectory of the ecosystem may: (1) follow natural restoration though secondary succession; (2) be re-directed through ecological restoration, or (3) be unattainable. In order to address the gaps in knowledge about restoration and recovery of estuarine and coastal ecosystems, this special feature includes the present overview and other contributions to provide a synthesis of our knowledge about recovery patterns, rates and restoration effectiveness. From the 51 examples collated in this contribution, we refine the recovery from the list of stressors into six recovery mechanisms: (1) recovery from sediment modification, which includes all aspects of dredging and disposal; (2) recovery by complete removal of stressors limiting natural ecosystem processes, which includes tidal marsh and inundation restoration; (3) recovery by speed of organic degradation, which includes oil discharge, fish farm wastes, sewage disposal, and paper mill waste; (4) recovery from persistent pollutants, which includes chemical discharges, such as TBT; (5) recovery from excessive biological removal, related to fisheries and (6) recovery from hydrological and morphological modification. Drawing upon experience both from these many examples and from an example of one comprehensive study, we show that although in some cases recovery can take <5 years, especially for the short-lived and high-turnover biological components, full recovery of coastal marine and estuarine ecosystems from over a century of degradation can take a minimum of 15–25 years for attainment of the original biotic composition and diversity may lag far beyond that period.

Keywords Restoration · Recovery · Long-term data series · Ecological integrity · Marine and estuarine ecosystems

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Introduction

The coastal zone is subject to many and varied changes resulting from human activities and natural processes (Aubry and Elliott 2006), which can impair the health and fitness of resident biota (Adams 2005) as well as the ability of the coastal zone to deliver ecosystem functions and the goods and services for human well-being (Beaumont et al. 2007; Costanza et al. 1997). In recent times, many estuarine and coastal marine ecosystems have experienced increasing degradation (Halpern et al. 2008a). This degradation can be caused by multiple stressors, including hydromorphological and sediment barriers (e.g. dams), toxic chemical pollutants, excess nutrient inputs, hypoxia, turbidity, suspended sediments and other ecosystem alterations, which can impact resources through single, cumulative or synergistic processes (Adams 2005; McLusky and Elliott 2004). Ecosystem degradation and pollution problems are correlated with increases in population density (Dauer et al. 2000) and significant efforts to monitor estuaries have been developed to assess their status (Latimer et al. 2003). However, within monitoring networks there is the need to separate catchment and diffuse stressors from localised and point-source problems. As an underlying philosophy, the DPSIR (drivers–pressures–state change–impact–response) approach (adapted from OECD 1993) has been widely used and accepted (see Borja and Dauer 2008) wherein each of the main drivers of change create a set of pressures; in turn, each pressure creates a set of state changes in the natural environment, which leads to impacts on the human system; these then require a set of responses, often involving legislative or economic instruments, which can control the excesses and thus lead to the adoption of the ecosystem approach stricto sensu (e.g. Mee et al. 2008). Through monitoring networks, establishing the relationships between drivers, their pressures (stressors) and impacts (effects) provides managers with a scientific basis upon which to build a consensus on solutions to problems (Latimer et al. 2003).

Anthropogenic pressures altering natural ecosystems are largely the result of societal and economic development (Borja and Dauer 2008). Natural ecosystems may recover from anthropogenic perturbations when secondary succession returns the ecosystem to the pre-existing condition or state. However, depending upon the scales of time, space and intensity of perturbations (sensu Connell and Slatyer 1977), return along the historic trajectory of the ecosystem may: (1) follow natural restoration though secondary succession; (2) be re-directed through ecological restoration (sensu SER 2002), where secondary succession is assisted by anthropogenic intervention (Aronson and Le Floc'h 1996b; Elliott et al. 2007; Halpern et al. 2008b; Simenstad et al. 2006; Stein and Cadien 2009) to produce ecosystems that are resilient to normal periodic stress and are as self-sustaining as reference ecosystems; or (3) be unattainable (Duarte et al. 2009). Despite this, ecological engineering may be used, for example by rehabilitating wetlands, to produce a sustainable ecosystem (i.e. one that maintains ecosystem processes while at the same time maintaining ecosystem services to deliver societal benefits such as producing fisheries or improving water quality).

In the latter case, through ecological engineering, a sustainable ecosystem may be developed that integrates human society with its natural environment for the benefit of both (Bergen et al. 2001; Lewis 2005). However, such a mutually beneficial and sustainable ecosystem will be inherently different from the historic ecosystem and the historic environmental homeostasis, defined as the inherent variability and resilience in the ecosystem required to mitigate or buffer anthropogenic change (see Elliott and Quintino 2007), will never be re-attained. Hence, legislation worldwide, such as the Clean Water Act, in the USA, or the Water Framework Directive (WFD) and the Marine Strategy Framework Directive, in Europe, includes restoration of degraded aquatic habitats as one of the primary goals (Apitz et al. 2006; Borja et al. 2008a). One of the most innovative aspects of this legislation is to base management decisions on the ecological effects of pressures, acknowledging that sensitivity and resilience vary substantially across ecosystems. This presents the challenge to translate data on the structure and dynamics of biotic communities into information for designing effective restoration (Hering et al. 2010). The rate of ecosystem recovery and the associated controlling factors are particularly difficult to identify, as exemplified by the paucity of literature on this topic. Without long-term data from the analysis of biological and chemical indicators, and an understanding of historical human activities, it is difficult to assess where an ecosystem is positioned along a trajectory to recovery (Latimer et al. 2003).

Although the response of different organisms to restoration activities is well known in some cases (Elliott et al. 2007), there is a lack of empirical data on relevant spatial and temporal scales over which restoration occurs (Connell and Slatyer 1977) and on the relationships among the patterns and rates of biotic and physico-chemical changes. For example, Jones and Schmitz (2009) developed a broad overview of time scales required for recovery across both terrestrial and aquatic ecosystems and found average recovery times of 10 to 20 years for brackish and marine systems. However, these authors emphasised that pre-disturbance data were available for only 20% of the reviewed studies, a factor that rendered the assessment of recovery in 80% of the studies subjective. While the literature shows that there are few studies which show the trajectory for decline, even fewer provide descriptions and mechanisms for the recovery.

There are very few examples of long-term monitoring data, including different biological elements (i.e. plankton, benthos, fishes, etc.) together with physico-chemical data from waters and sediments, showing the recovery trajectories after remediation or restoration processes in marine environments (see examples in Borja et al. 2009a; Elliott et al. 2007; Jones and Schmitz 2009; Lotze et al. 2006; Simenstad et al. 2006; Stein and Cadien 2009; and Yuksek et al. 2006).

To address this fundamental gap in our knowledge about recovery patterns and rates in estuarine and coastal ecosystems, in January 2009, we organised a session on "Medium and long-term recovery of marine and estuarine systems—a guide to providing useful information in new scenarios to restore ecological integrity" for the American Society of Limnology and Oceanography (ASLO) Conference in Nice (France). This special feature includes some of the presentations in this session, with the purpose of providing a useful synthesis for systematising the state of knowledge about recovery patterns, rates and effectiveness after restoration. Moreover, we include an example of an estuarine ecosystem, summarising our knowledge in long-term recovery of aquatic systems, of multiple biological and physico-chemical elements. Our ultimate goal, both in the meeting and this publication, is to provide data and opportunities to model scenarios of the assessment of ecological potential and give lessons for recovery.

Recovery of Marine Ecosystems After Removing Different Human Pressures

We surveyed the current literature and identified 51 long-term cases (Table 1) where (1) actions were taken to remove or reduce human pressure effects; (2) information on the responses of biological elements was available, and (3) medium or long-term monitoring of the recovery occurred. These case studies are from 23 different anthropogenic pressures upon various biological elements (macroinvertebrates, meiofauna, macroalgae, fishes and angiosperms) and include different geographical regions (19 countries from all continents), as well as different substrata and tidal levels. The time span of recovery after removal of the pressure is highly variable, extending from several months (in the case of meiofauna) to more than 22 years (in hard-bottom macroalgae and some sea grass species).

Severe impacts, whether acute, such as large oil spills, chronic (low level inputs) or persistent over time and space (such as sewage sludge disposal, extensive wastewater discharge or mine tailings), require periods up to 10–25 years for complete recovery (see Table 2, derived from Table 1). Conversely, restoration after physical disturbance (including dredging and restoration of tidal inundation) that does not leave a "legacy" stressor such as a persistent contaminant can take 1.5–10 years for recovery, although some sensitive organisms (such as angiosperms) may take over 20 years to recover (Table 2). Fish assemblages appear to recover from most pressures in less than 10 years, although it may take several decades to acquire a full species complement after starting from a state without any fish community, as was observed in the 1960s in the Thames River estuary, London, UK (McLusky and Elliott 2004). Although the data on speed of recovery need further interrogation, current analyses indicate that, if the physico-chemical system is restored, then colonisation, and hence recovery, will occur if there are sufficient recruits available. Again we need to determine whether the rate and pattern of recovery can be linked to the turnover and/or life span of the organisms (e.g. meiofaunal recovery is faster than macrofaunal) and also to the relationship between the timing of the cessation of the stressor and the recruitment and influx of organisms. For example, if sediment disturbance ceases before the spring–summer influx of recruits in temperate zones, then restoration of the community may be faster than if the influx occurred in other seasons. Similarly, there is the need to assess whether there are differences in the speed of recovery by organisms with planktonic dispersal (e.g. many macrobenthic species) from those with vegetative reproduction (e.g. sea grasses; Mazik et al. 2007).

In a few cases, recovery was not at all evident. From four well-studied coastal ecosystems, Duarte et al. (2009) did not observe a return in simple biological variables (such as chlorophyll *a* concentration) following the assumed reduction of nutrient loads during two decades. In the Chesapeake Bay, despite extensive restoration efforts (including point-source reductions, fisheries management, sea grass plantings and oyster bed restoration), nutrient concentrations and associated ecological health-related water quality and biotic metrics have generally shown little improvement and, in some cases, large decreases since 1986 (Williams et al. this issue), keeping the submersed aquatic vegetation coverage below restoration targets (Orth et al. 2010). This may be reflected by the hysteresis term in the model proposed by Elliott et al. (2007) which indicates that the trajectory of degradation may be different from the trajectory of recovery; that difference can be regarded as a degree of 'memory' in the system (Peterson 2002) which may be related to the type of stressor and the ability of it to be assimilated (see below).

The data collated in Table 1 do not indicate geographical patterns in recovery. For the same pressure, the response is similar in most locations and small differences are related to the turnover time in the system or in the dominant and bio-engineering species. Consequently, we would not expect a geographical pattern if the mechanisms of recovery are generic and relate to the functioning of the systems rather than the species identities which may vary geographically.

When the restoration examples are organised by the type of stressor and the organisms studied during the recovery (Table 2), we find the stressors can be associated with one of six recovery mechanisms (Table 3): (1) recovery from sediment modification, which includes all aspects of dredging and disposal; (2) recovery by habitat creation,

Table 1 Long-term monitoring of different anthropogenic pressures, worldwide, in different substrata and tidal levels, using different biological elements, showing the time span of recovery after restoration or removing pressure

Pressure	Location	Substrata	Intertidal/ subtidal	Biological elements	Time for recovery	Authors
Sewage sludge disposal	Northumberland coast (UK)	Soft	Subtidal	Macroinvertebrates	>3 years	Birchenough and Frid 2009
Sewage sludge disposal	Garroch Head (Firth of Clyde, Scotland)	Soft	Subtidal	Macroinvertebrates	Incomplete after 14 years	Moore and Rodger 1991
Sewage sludge disposal	Liverpool Bay (UK)	Soft	Subtidal	Macroinvertebrates	Incomplete after 5 years	Whomersley et al. 2007
Wastewater discharge	California (USA)	Soft	Subtidal	Macroinvertebrates	18 years	Stein and Cadien 2009
Wastewater discharge	Boston harbour (USA)	Soft	Subtidal	Macroinvertebrates	10–15 years	Diaz et al. 2008
Wastewater discharge	Basque estuaries (Spain)	Soft	Subtidal	Macroinvertebrates	10–15 years	Borja et al. 2006, 2009b
Wastewater discharge	Marseille (France)	Soft	Subtidal	Macroinvertebrates	>7 years	Bellan et al. 1999
Wastewater discharge	Basque coast (Spain)	Soft/hard	Subtidal	Invertebrates and algae	>6 years	Borja et al. 2009b
Wastewater discharge	Abra of Bilbao (Spain)	Hard	Intertidal	Macroalgae	Incomplete after 22 years	Díez et al. 2009
Wastewater discharge	Basque estuaries (Spain)	Soft	Subtidal	Fishes	3–10 years	Uriarte and Borja 2009
Wastewater discharge	Tagus estuary (Portugal)	Soft	Intertidal/ subtidal	Macroinvertebrates	Incomplete after 12 years	Chainho et al. this issue
Eutrophication	Victoria Harbour, Hong Kong	Soft	Subtidal	Macroinvertebrates	>3 years	Shin et al. 2008
Eutrophication	Orbetello lagoon (Italy)	Soft	Subtidal	Macroinvertebrates	>6 years	Lardicci et al. 2001
Eutrophication	Mondego estuary (Portugal)	Soft	Intertidal	Zostera noltii and macroinvertebrates	>4 years	Dolbeth et al. 2007; Neto et al. this issue
Eutrophication	Tampa Bay (Florida, USA)	Soft	Subtidal	Sea grasses	Incomplete after 20 years	Greening and Janicki 2006
Oxygen depletion	Gullmarsfjord (Sweden)	Soft	Subtidal	Macroinvertebrates	2 years	Rosenberg et al. 2002
Oil spill	Various	Soft/hard	Intertidal/ subtidal	Various	2–10 years	Kingston 2002
Oil-refinery discharge	Barbadun estuary (Spain)	Soft	Intertidal	Macroinvertebrates	2–3 years	Borja et al. 2009b
Oil-refinery discharge	Milford Haven (UK)	Hard	Intertidal	Macroinvertebrates	2–3 years	Wake 2005
Oil-refinery discharge	Barbadun estuary (Spain)	Soft	Subtidal	Fishes	2–3 years	Uriarte and Borja 2009
Fish farm	Archipelago Sea (Finland)	Soft	Subtidal	Macroinvertebrates	Incomplete after 7 years	Kraufvelin et al. 2001
Fish farm	Hornillo Cove (Mediterranean, Spain)	Soft	Subtidal	Macroinvertebrates	2–3 years	Sanz-Lázaro and Marín 2006
Fish farm	Tasmania (Australia)	Soft	Subtidal	Macroinvertebrates	>2.5 years	Macleod et al. 2008
TBT	Crouch Estuary, Essex (UK)	Soft	Subtidal	Macroinvertebrates	3–5 years	Smith et al. 2008
Mine tailings	Rupert Inlet, British Columbia (Canada)	Soft	Subtidal	Macroinvertebrates	4–15 years	Burd 2002
Mine tailings	Affarlikassa and Quaamarujuk (Greenland)	Soft	Subtidal	Macroinvertebrates	>15 years	Josefson et al. 2008
Pulp mill	Swedish fjord	Soft	Subtidal	Macroinvertebrates	6–8 years	Rosenberg 1972, 1976
Physical disturbance	South Africa	Hard	Intertidal	Macroinvertebrates	3 years	Dye 1998
Physical disturbance	Peru Basin	Soft/hard	Deep sea	Megafauna	Incomplete after 7 years	Bluhm 2001
Land claim	Bidasoa estuary (Spain)	Soft	Intertidal	Macroinvertebrates	2 years	Marquiegui and Aguirrezabalaga 2009
Land claim	Nakdong River estuary (Korea)	Soft	Subtidal	*Zostera marina*	Incomplete after 20 years	Park et al. 2009
Marsh restoration	Delaware Bay (USA)	Soft	Subtidal	Fishes	1–2 years	Able et al. 2008
Marsh and tidal restoration	Long Island Sound (USA)	Soft	Intertidal/ subtidal	Vegetation, macroinvertebrates, fishes, birds	5–20 years	Warren et al. 2002
Realignment of coastal defences	Tollesbury, Essex (UK)	Soft	Intertidal	Marshes and macroinvertebrates	>6 years	Garbutt et al. 2006
Lagoon isolation	East Harbor, Massachusetts (USA)	Soft	Subtidal	Molluscs	Incomplete after 3 years	Thelen and Thiet 2009
Lagoon isolation	Lake Veere (Netherlands)	Soft	Intertidal/ subtidal	Macroinvertebrates	Incomplete alter 4 years	Wijnhoven et al. this issue
Dyke and marina construction	Oria estuary (Spain)	Soft	Intertidal	Macroinvertebrates	2 years	Borja et al. 2009b
Dyke and marina construction	Oria estuary (Spain)	Soft	Subtidal	Fishes	2–3 years	Uriarte and Borja 2009
Dredging and sediment disposal	Basque coast and estuaries (Spain)	Soft	Subtidal	Macroinvertebrates	2–3 years	Borja et al. 2009b

Table 1 (continued)

Pressure	Location	Substrata	Intertidal/ subtidal	Biological elements	Time for recovery	Authors
Dredging and sediment disposal	Mecklenburg Bay (western Baltic Sea)	Soft	Subtidal	Macroinvertebrates	2 years	Powilleit et al. 2006
Dredging and sediment disposal	Mississippi sound (USA)	Soft	Subtidal	Macroinvertebrates	2 years	Wilber et al. 2007
Dredging	Worldwide	Soft	Intertidal/ subtidal	Sea grasses	2–>5 years	Erftemeijer and Robin Lewis 2006
Dredging	Basque estuaries (Spain)	Soft	Subtidal	Fishes	2–3 years	Uriarte and Borja 2009
Aggregate dredging	Hastings Shingle Bank (UK)	Soft	Subtidal	Macroinvertebrates	3–4 years	Cooper et al. 2008
Aggregate dredging	South East England (UK)	Soft	Subtidal	Epifauna	2–3 years	Smith et al. 2006
Sand extraction	Coast of Ravenna (Adriatic Sea, Italy)	Soft	Subtidal	Macroinvertebrates	2–4 years	Simonini et al. 2007
Sediment disposal	Hamford Water, Essex (UK)	Soft	Intertidal	Meio and macrofauna	3–18 months	Bolam et al. 2006
Sediment disposal	Laguna Madre, Texas (USA)	Soft	Subtidal	Sea grass, Macroinv., Fishes	>5 years	Sheridan 2004
Sediment disposal	Chesapeake Bay (USA)	Soft	Subtidal	Macroinvertebrates	1.5 years	Schaffner this issue
Fish trawling	North Sea	Sand-gravel	Subtidal	Macroinvertebrates	2.5–7 years	Hiddink et al. 2006
Fish trawling	North Sea	Sand–gravel	Subtidal	Fishes	5–10 years	Maxwell and Jennings 2005

Table 2 Summary of time for recovery, for different biological elements and substrata, under different pressures

Pressure	Substrata	Intertidal/subtidal	Biological elements	Time for recovery
Sediment disposal	Soft	Intertidal	Meio and macrofauna	3–18 months
Marsh restoration	Soft	Subtidal	Fishes	1–2 years
Oxygen depletion	Soft	Subtidal	Macroinvertebrates	2 years
Land claim	Soft	Intertidal	Macroinvertebrates	2 years
Oil-refinery discharge	Soft/hard	Intertidal/subtidal	Macroinvertebrates, fishes	2–3 years
Dyke and marina construction	Soft	Intertidal/subtidal	Macroinvertebrates, fishes	2–3 years
Lagoon isolation	Soft	Subtidal	Molluscs	>3 years
Aggregate dredging	Soft	Subtidal	Macroinvertebrates, epifauna	2–4 years
TBT	Soft	Subtidal	Macroinvertebrates	3–5 years
Dredging	Soft	Intertidal/subtidal	Sea grasses, macroinvertebrates, fishes	2–>5 years
Sediment disposal	Soft	Subtidal	Sea grass, macroinvertebrates, fishes	>5 years
Eutrophication	Soft	Subtidal	Macroinvertebrates	>3–>6 years
Realignment of coastal defences	Soft	Intertidal	Marshes and macroinvertebrates	>6 years
Fish farm	Soft	Subtidal	Macroinvertebrates	2–>7 years
Physical disturbance	Soft/hard	Intertidal/deep sea	Macroinvertebrates, megafauna	3–>7 years
Pulp mill	Soft	Subtidal	Macroinvertebrates	6–8 years
Oil spill	Soft/hard	Intertidal/subtidal	Various	2–10 years
Fish trawling	Sand–gravel	Subtidal	Macroinvertebrates, fishes	2.5–10 years
Wastewater discharge	Soft	Subtidal	Fishes	3–10 years
Sewage sludge disposal	Soft	Subtidal	Macroinvertebrates	3–>14 years
Mine tailings	Soft	Subtidal	Macroinvertebrates	4–>15 years
Marsh and tidal restoration	Soft	Intertidal/subtidal	Vegetation, fishes, birds	5–20 years
Wastewater discharge	Soft	Subtidal	Macroinvertebrates, sea grasses	7–20 years
Land claim	Soft	Subtidal	*Zostera marina*	>20 years
Wastewater discharge	Hard	Intertidal	Macroalgae	>6–>22 years

Table 3 Mechanisms of recovery and for recovery

Mechanisms for recovery	Recovery features	Recovery
From sediment modification	Usually in areas of high sediment turnover and sediment influx, with or without organisms colonising	A function of the ease of sediment influx and the organism influx
By habitat creation	Create the appropriate physical environment and then allow organisms to colonise	A function of the ease of creating the suitable space and the ease of influx of organisms
By organic matter degradation and reduction of nutrient load	Recovery occurs once the excess organic matter is broken down (in the case of sewage and oil), any toxic pollutants have evaporated (from oil spills), and the excess of nutrients is removed; this is more difficult in fine sediments than coarse sediments and in low-energy areas than in high energy areas	A function of the original amount of organic matter stored in the system and the conditions for its breakdown; shown by an absence of symptoms of eutrophication (algal blooms, oxygen depletion, etc.)
From persistent pollutants	The ability of the system to sequester/bury the persistent pollutants or disperse them to reach low background levels	A function of the original amount and toxicity of the pollutants, their degradation potential by physical, chemical or biological methods and thus the speed of sequestration
From excessive biological removal	The ability of the system either to replenish stocks naturally or with human interference through restocking	A function of the severity of the biological removal (over-fishing) and the rate of recolonisation/recruitment and reproduction
From hydrological–morphological modification	The ability to remove barriers and restore water flow, current patterns, salinity balance, etc.	A function of the ease with which these hydromorphological conditions can be restored naturally or with human interventions

which includes marsh restoration, restoration of tidal inundation, etc.; (3) recovery by speed of organic degradation and reduction of nutrient load, which includes oil discharges and oil spillages, fish farms, sewage disposal, paper mill waste, etc.; (4) recovery from persistent pollutants, which includes chemical discharges, TBT, etc.; (5) recovery from excessive biological removal, which relates to fisheries and (6) excessive water abstraction had occurred and so required recovery from hydrological and morphological modification, which refers to all cases where physical barriers were in place (which also affects the salinity balance) and habitats previously were isolated. Hence, we emphasise that the pattern of the recovery (Table 3) is often a function of anthropogenic interference to the physical system and then allowing either natural sediment influx or the influx of recruits to start re-building the community structure. Once the community structure has been created (due to organisms colonising available or created niches) the community functioning (including inter- and intra-specific interactions such as predator–prey relationships and competition) will develop (Gray and Elliott 2009). In many instances (Tables 1 and 2) the studies focus on an initial reappearance of particular biological elements but we caution that the presence of a biological element following colonisation is not necessarily an indication that a fully functioning ecosystem has been created (Mander et al. 2007; Mazik et al. 2007). For example, the recolonisation by one group of organisms (e.g. predators), will not create a sustainable ecosystem if the conditions are not suitable for the recolonisation by interacting species (e.g. their prey).

Long-term Recovery in an Estuarine System: the Nervión River Estuary as Example

Very few of the studies indicated above consider more than one ecological component. Therefore to better illustrate the way in which the restoration of ecosystem integrity has been determined by an effective monitoring programme we present an exemplary case study of a long-term physico-chemical and biological (i.e. plankton, benthos, fishes, seabirds) data set, together with information on historical human activities from the Nervión estuary (Basque Country, northern Spain). The hydrogeomorphology of the estuary was modified dramatically by urban, industrial and port settlement; these modifications include almost the whole of the original estuary, during the last 150 years (Cearreta et al. 2004). Exploitation of locally abundant iron ore led to the early industrial development of the area in the mid-nineteenth century, coincident with an increase in the density of the population (to c. 2,250 inhabitants per square kilometre and the present total of 1 million inhabitants) (Belzunce et al. 2004). As a consequence, over the last 150 years the estuary has received wastes from many sources (e.g. mineral sluicing, industrial wastes and urban effluents) that have significantly degraded its environmental quality (Belzunce et al. 2001, 2004; Borja et al. 2008b). Before treatment in 1984, the estuary received some 250,000 m^3 per day of urban wastewaters and 67,000 m^3 per day of industrial waters (produced mainly by the chemical, iron–steel and paper sectors) that were highly contaminated with toxic products (cyanide, heavy metals, fluorides and phenols (Borja et al. 2006)). This produced extremely low concentrations of dissolved oxygen (even anoxia) in the water column and a high content and concentration of organic matter and heavy metals in the sediments (Belzunce et al. 2001, 2004) as well as deteriorating benthic communities (Gorostiaga et al. 2004).

In a major initiative to reverse this situation of poor environmental quality, in 1979 the *Consorcio de Aguas Bilbao-Bizkaia*, the Local Authority responsible for water supply and wastewater treatment, approved a sewage treatment project that initiated an estuary-scale recovery (Fig. 1). Diversion of discharges to a water treatment plant began in the late 1980s and the physico-chemical water treatment began in 1990 (Franco et al. 2004); in 1995, the company causing most pollution in the estuary (the iron and steel industry, *Altos Hornos de Vizcaya*), was closed and, in 2001, a secondary treatment plant came into operation (Borja et al. 2006; Fig. 1). This has resulted in a progressive improvement in the physico-chemical properties (García-Barcina et al. 2006), benthic (Borja et al. 2006, 2009b; Bustamante et al. 2007; Díez et al. 2009) and fish assemblages (Uriarte and Borja 2009; Fig. 1).

However, although oxygen conditions have increased significantly during the clean-up process (Borja et al. 2006, 2009b), the estuary continues to be moderately polluted by priority substances (i.e. metals and organic compounds) in waters (Fernández et al. 2008), sediments (Bartolomé et al. 2006; Belzunce et al. 2001; Prieto et al. 2008) and biota (Besada et al. 2008; Franco et al. 2002; Fig. 1). Although the concentrations have decreased in recent years both in waters and sediments (Leorri et al. 2008; Tueros et al. 2009), the presence of these substances can affect biological elements when they recolonise previous azoic areas (Borja et al. 2006).

From this published information, the history of the recovery within the Nervión River estuary can be reconstructed following the main restoration milestones (Fig. 1). We identify a sequence of four phases:

1. When the physics and chemistry of the system were restored, there was a progressive increase of dissolved oxygen and a reduction of pollutants in waters and sediments.
2. The biological elements colonised the inner part of the estuary, probably initially through the plankton and then by mobile soft-bottom macroinvertebrates coincident with a progressive increase in richness and diversity and a decrease in AMBI values (AZTI's Marine Biotic Index, an indicator of the proportion of sensitive/opportunistic species; Borja et al. 2000). This recovery of soft-bottom macroinvertebrates has progressed over at least 15 years. In coastal waters, although apparently less affected by pollution, recovery of hard-bottom macroalgae took at least 14 years, showing that recovery of structural species was much slower than the remainder of species.

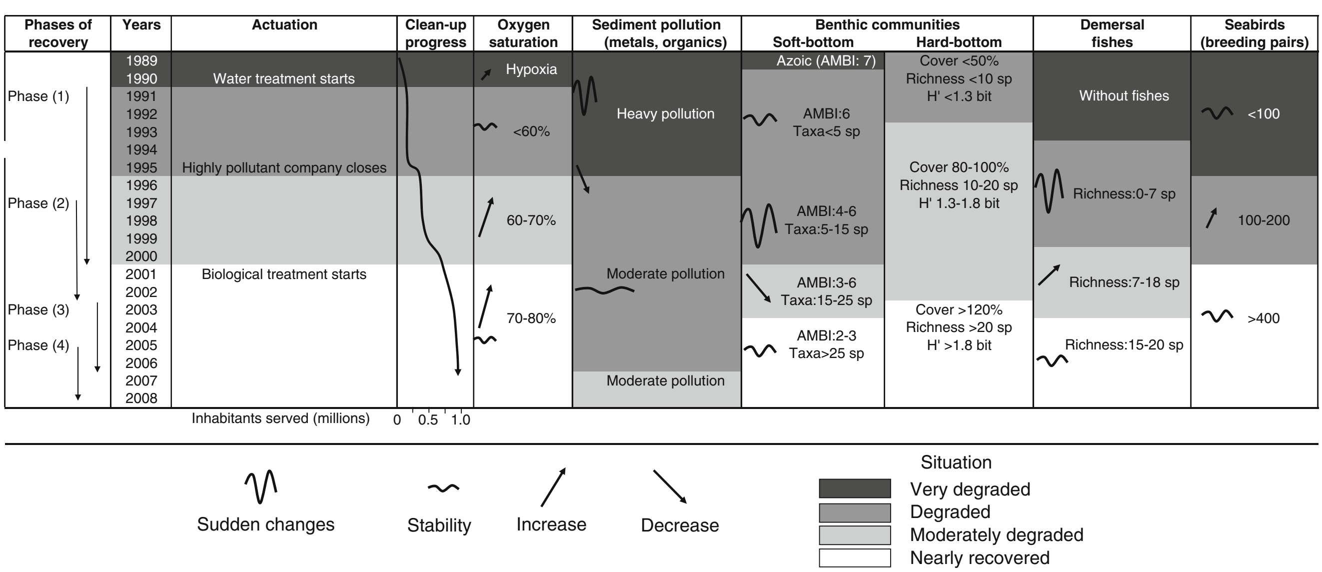

Fig. 1 Recovery patterns and recovery phases (see text) of the quality within the Nervión estuary (Basque Country, northern Spain), during a period of water treatment progression. Data source: clean-up progress (García-Barcina et al. 2006), oxygen and sediment (Borja et al. 2006, 2009b; Tueros et al. 2009; Javier Franco (AZTI-Tecnalia), personal communication), estuarine soft-bottom macroinvertebrates (Borja et al. 2006, 2009b), coastal macroalgae from hard-bottom substrata (Díez et al. 2009), demersal fishes (Uriarte and Borja 2009), seabirds (as breeding pairs of *Phalacrocorax carbo*, Javier Franco (AZTI-Tecnalia) personal communication). *AMBI* AZTI's Marine Biotic Index (Borja et al. 2000), *H′* Shannon–Wiener diversity index

3. More complex biological interactions and functions began developing, as indicated by the demersal fishes which started to colonise the inner part of the estuary at least 4 years after the initial recovery of soft-bottom macroinvertebrates. Although this pattern of recovery is probably related to the changes in oxygen concentration in bottom layers, it also coincides with sufficient food availability to support permanent colonisation of these inner estuary locations. After the start of colonisation, however, the near complete recovery was achieved in 10 years, probably due to the well-known contribution of marine species to the total species richness within the estuaries (Nicolas et al. 2010). We emphasise that recovery from temporary ecosystem loss, such as water quality problems (including anoxia), may take less time than recovery from permanent ecosystem loss (such as land claim), where the appropriate lost ecosystem needs to be created (also see Elliott et al. (2007) and references therein). Finally, seabirds (in this particular case, the number of breeding pairs of the cormorant *Phalacrocorax carbo*) started to increase in number 2 years after the initial recovery of fishes in the inner part of the estuary. These seabirds, which feed on fishes, showed a rapid recovery after completion of the clean-up.
4. Recovering biological communities will start influencing the physico-chemical system through bioturbation, biosedimentation, vegetative growth, etc. (see Gray and Elliott 2009).

In spite of this recovery, vigilance and monitoring is necessary, given the continued increase in population that may counterbalance the management efforts in the longer term, thereby constituting an additional trajectory to the three presented in the previous section.

The case study of the Nervión River estuary conforms to the conceptual model of changes to the system status described by Bricker et al. (2007) and Elliott et al. (2007) commensurate with the increasing and relocated wastewater discharge volume and decreasing pressure due to improved wastewater treatment (Fig. 2). Although the available information is somewhat incomplete, foraminiferal and sediment data (Cearreta et al. 2004) show that before mid-nineteenth century the estuary was in good status (sensu the WFD, see Borja et al. 2009b). In the second half of the nineteenth century, after land claim, dredging and industri-

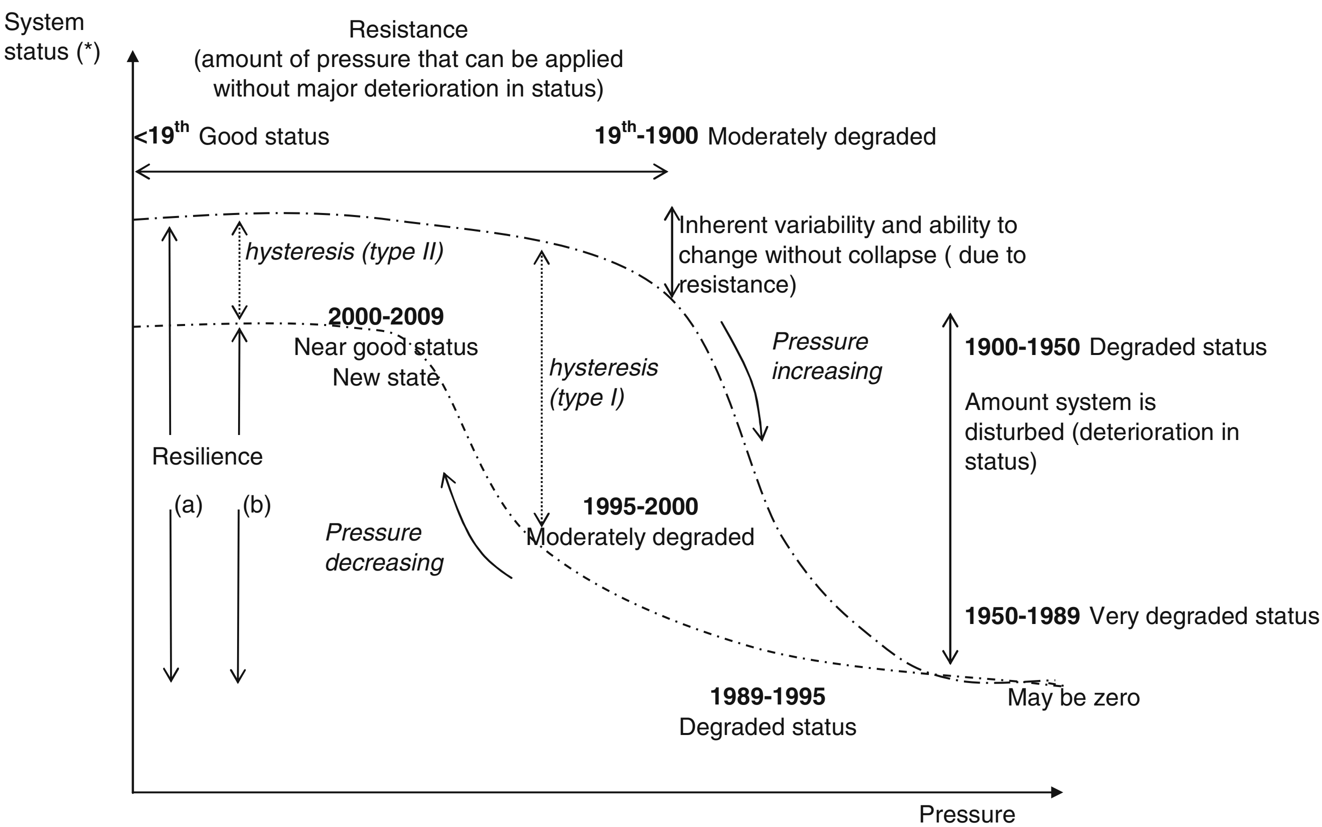

Fig. 2 A conceptual model of changes to the state of Nervión estuary (Basque Country, northern Spain) with increasing (increasing wastewater discharge volume) and decreasing (wastewater treatment improvement) pressure (adapted from Elliott et al. (2007)). (*a*) Complete resilience; (*b*) incomplete resilience

alisation, the estuary rapidly degraded through the first half of the twentieth century and until the end of the 1980s (Fig. 2). Recovery does occur with decreasing pressures, in part by wastewater treatment, but it is uncertain whether it will be complete because many ecosystems have been reduced or lost (e.g. intertidal areas, salt-marshes, etc.). The net result will probably lead to a decreased abundance, richness and biomass of some biological taxa. Hence, as shown by Elliott et al. (2007) and Tett et al. (2007), an inherent hysteresis in the system may be unquantified (see Fig. 2), providing a new state for the Nervión estuary, whose designation was changed to a Highly Modified Water Body (i.e. a water body with strong geomorphological changes, such as a harbour), after the WFD (Borja and Elliott 2007). Such hysteric trajectories can also be referred to as ecological regime shifts (sensu Carpenter and Brock 2006; Contamin and Ellison 2009) that could ultimately return to the 'alternative steady state' or remain on a trajectory toward 'modified' or entirely 'new' ecosystems (Aronson and Le Floc'h 1996a; Hobbs and Norton 1996). The analysis of this case study reinforces the conclusion that effective efforts at recovery require an a priori understanding that recovery rates will vary based upon the interaction of the type of anthropogenic pressure and specific biotic components, as well as post-hoc measurement of the trajectory of change.

Conclusions

The analysis here shows that although in some cases recovery can take <5 years, the full recovery of many coastal marine and estuarine ecosystems can take a minimum of 15–25 years from over a century of degradation and attainment of the original biotic composition and diversity and complete functioning may lag far beyond that, possibly at least another 25 years. Some ecosystems may never attain the technical definition of being restored, but end up irreversibly in an alternative state, as shown in the Nervión estuary. The recovery may achieve ecosystem structure, as indicated by the presence of appropriate organisms, but this does not necessarily mean that ecosystem functioning has been regained. Where restoration measures can be implemented rapidly, and natural processes fully recovered, there will in many cases be significant improvements of ecological status within this time span, although not necessarily to reach the original historical state (Hering et al. 2010). We also emphasise that we need to agree upon the restoration goals for the system and also what criteria will be used to determine attainment of the desired system (Simenstad et al. 2006). For example, we must question whether the system restored merely contains the structural elements, i.e. the appropriate species complement, or whether we are sufficiently competent in restoration to achieve a fully functioning system with the appropriate amount of primary production, predator–prey relationships and competition between species. Secondly, we need to determine whether dynamic marine systems, i.e. of highly variable hydrodynamics in open systems, recover more quickly than non-dynamic low-energy ones, such as accreting mudflat areas. We also emphasise that open systems cannot be restored other than by removing the stressor and allowing natural processes to aid the recovery. In contrast, we now have a large amount of experience in estuarine and coastal areas of helping systems recover some significant degree of functioning, even if this is not the same as the original system under pre-anthropogenic influences.

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