CHAPTER

11 Trade-Offs Between Biodiversity Conservation and Nutrients Removal in Wetlands of Arid Intensive Agricultural Basins: The Mar Menor Case, Spain

Julia Martínez-Fernández[a,*], Miguel-Angel Esteve-Selma[a], Jose-Miguel Martínez-Paz[b], María-Francisca Carreño[a], Javier Martínez-López[a], Francisco Robledano[a], Pablo Farinós[a]

[a]*Department of Ecology and Hydrology, University of Murcia, Murcia, Spain*

[b]*Department of Applied Economics, University of Murcia, Murcia, Spain*

[*]*Corresponding author: e-mail address: juliamf@um.es*

11.1 INTRODUCTION

Although wetlands provide a wide variety of ecosystem services such as biomass production, carbon storage, biodiversity, fish production, and nutrient removal (Okruszko et al., 2011), these ecosystems remain insufficiently valued in areas where they do not supply direct market goods, as is the case of many wetlands (Barbier, 2011; Maltby and Acreman, 2011), particularly those of arid and semi-arid environments. In this context, the concept of ecosystem services and their assessment may help to communicate the need for their conservation and sustainable management to policymakers and end-users (Maes et al., 2012).

In the last years, considerable efforts have been devoted to the assessment of ecosystem services at very different spatial, temporal, and conceptual resolutions using a variety of approaches, including monetary valuations of real or virtual markets for ecosystem goods and services, mapping of ecosystem services using remote-sensing monitoring data and modeling tools, among others. Model development usually constitutes a costly process in terms of required time and effort. Therefore, it is pertinent to ask whether modeling tools are useful for the assessment of ecosystem services, how they contribute to such an assessment, in what cases models are particularly needed, and what kind of models are more appropriate for this task.

In relation to the ecosystem services of wetlands, we consider that their assessment should adopt a perspective capable of accounting for the following issues:

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(i) the relationships between wetlands and their watershed; (ii) the drivers of change across time and how they affect the wetlands and their services; (iii) the identification of less common services and goods that might be important for specific wetlands; and (iv) the potential interactions among wetland ecosystem services. We try to illustrate the above-mentioned questions regarding the role of modeling tools on the assessment of ecosystem services, in light of these issues, with the case of the Mar Menor lagoon and associated wetlands, located in southeastern Spain, one of the most arid areas in Europe.

The Mar Menor lagoon is a hypersaline Mediterranean coastal lagoon located in southeast Spain. Ramsar Site since 1994, it is the largest water surface of the western Mediterranean coast (135 km^2 surface area) and is almost closed by a sand bar 22 km long. Inside the lagoon there are five volcanic islands. The lagoon is characterized by its hypersaline, clear, and relatively oligotrophic waters, with a low phytoplanktonic biomass, since primary production is dominated by macrophytes (Perez Ruzafa et al., 2002). Close to its internal shore is a series of coastal wetlands, Marina del Carmolí, Playa de la Hita, and Saladar de lo Poyo (Figure 11.1), described as coastal crypto-wetlands (Vidal-Abarca et al., 2003).

The Mar Menor lagoon and associated wetlands are important sites for wintering and breeding waterbirds (Martínez-Fernández et al., 2005; Esteve et al., 2008; Robledano et al., 2011). The lagoon and wetlands maintain 18 habitats of European interest, according to the Habitat Directive (92/43/ECC). The ecological value of the Mar Menor lagoon and associated wetlands has been recognized in a series of rules and resolutions, at regional, national, and international levels (Ramsar site, Special Protection Area for Birds, Site of Community Importance (SCI) and Special Protection Area for the Mediterranean).

The Mar Menor watershed is a 1270 km^2 plain slightly inclined toward the lagoon and drained by several ephemeral watercourses (ramblas), most of which only flow into the lagoon after big rainfall events. The area has a Mediterranean arid climate, with warm winters, an annual mean temperature about 17 °C, annual mean rainfall of 330 mm, and a high interannual rainfall variation. As in other Mediterranean watersheds, heavy rainfall events and flash-floods play a major role (David et al., 1997; Xue et al., 1998), leading to the mobilization of important stocks of nutrients stored in the watershed. More than 80% of the total area of Campo de Cartagena is used for agriculture, especially for open-air horticultural crops, citrus fruits, and greenhouses (Figure 11.2).

The Mar Menor watershed has experienced profound land use changes, that is, urban and tourist development and increased areas of irrigated lands due to the Tagus-Segura water transfer. These changes have led to a noticeable increase in water and nutrient flows reaching the lagoon and associated wetlands. In this chapter we analyze (i) to what extent and how the ecosystem services of wetlands of arid environments, particularly nutrients removal and biodiversity conservation, can be affected by the land use and the hydrological changes at watershed scale; and (ii) whether there are interactions (synergistic effects or trade-offs) among such ecosystem services.

To this aim, we have developed and applied a dynamic model of the Mar Menor watershed to estimate the inflow of nutrients from diffuse and point sources reaching

FIGURE 11.1

Location of Mar Menor wetlands. PH, Playa de la Hita; MC, Marina del Carmolí; LP, Saladar de Lo Poyo.

the lagoon and associated wetlands. Then, some effects on the lagoon dynamics (jellyfish outbreaks) are described. Next, we assess the effects on some important ecosystem services, first regarding nutrient removal and second regarding biodiversity conservation, by means of the aquatic bird assemblages and the natural habitats of wetlands. Then we apply the dynamic model and environmental economic techniques to assess and value the wetland ecosystem service of nutrients removal and to compare among different measures to reduce nutrient inflows into the lagoon. We discuss interactions and potential trade-offs between the two studied wetland

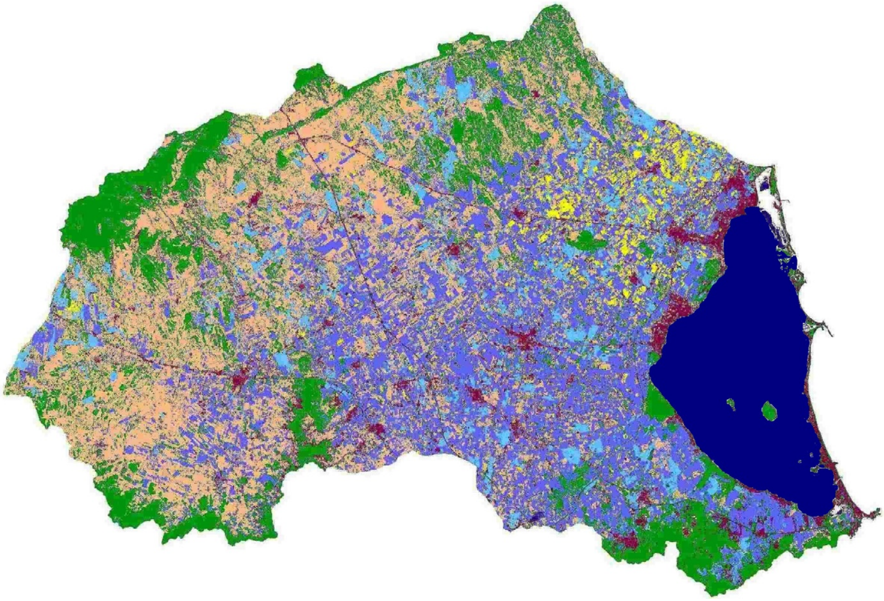

FIGURE 11.2

Mar Menor watershed showing agriculture as dominant land use.

ecosystem services (biodiversity conservation and nutrients removal) and finally make some concluding remarks on the role of models for the assessment of ecosystem services, particularly regarding wetlands.

11.2 DYNAMIC MODELING OF THE MAR MENOR WATERSHED

11.2.1 Model description

A watershed model has been developed with key environmental and socio-economic factors driving the dynamics of nutrient inputs into the lagoon (Chapelle et al., 2005; Martínez-Fernández and Esteve-Selma, 2007). It has a long-term time horizon, allowing the simulation of a 30-year time span on a daily basis. Several sectors have been considered: (i) nitrogen flows and compartments, accounting for N content in the soil solution, litterfall, live material, and humus; (ii) phosphorus flows and compartments, accounting for *P* content; (iii) land-use changes between natural areas, irrigated-tree crops, open-air horticultural crops, greenhouses, and urban areas; (iv) salty wastewater from desalination plants; (v) the role of wetlands on nutrient removal; (vi) nutrient inputs from urban sources; and (vii) the economic costs of main management measures. Information for model calibration, parameter estimation, and data inputs is provided by a range of sources, including empirical field studies, statistical databases, and literature. Figure 11.3 presents a simplified diagram of main model sectors.

The wetlands sector takes into account the active wetland area, the retention capacity for nitrogen and phosphorous, and the effect of water volume on nutrient retention. Empirical data from de Marina del Carmoli wetland were used to determine the retention ratio as a function of water volume (Figure 11.4) and the length of watercourse inside the wetlands.

The socioeconomic issues involved in the export of nutrients at watershed scale are not considered in a separate model or sector. Instead, they become part of the variables defining all model sectors, in close interaction with the environmental factors. The land use sector (Figure 11.5) takes into account the area and main land use changes between natural vegetation, drylands, urban areas, and each type of irrigated land (irrigated-tree crops, open-air horticultural crops, and greenhouses). The urban sector takes into account the resident population, the seasonal dynamics of tourist population, the efficiency of wastewater treatment plants, and the amount of wastewater reused for agriculture.

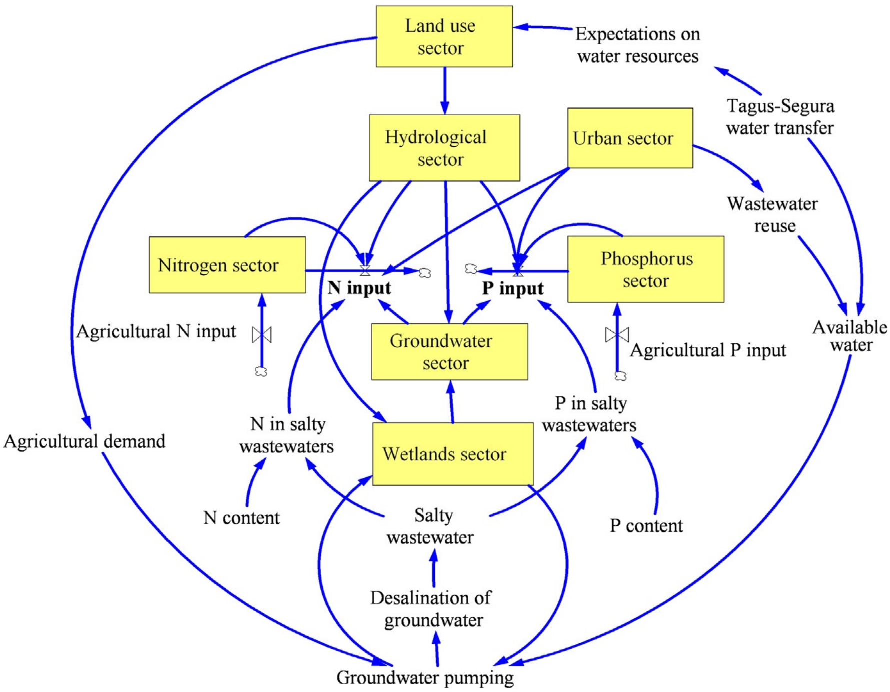

FIGURE 11.3

Simplified diagram of Mar Menor watershed model.

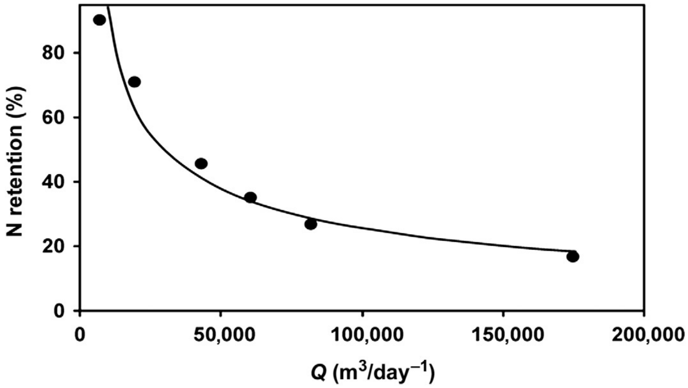

FIGURE 11.4

Nitrogen retention ratio as a function of total water inflow. Empirical data from Marina del Carmoli wetland.

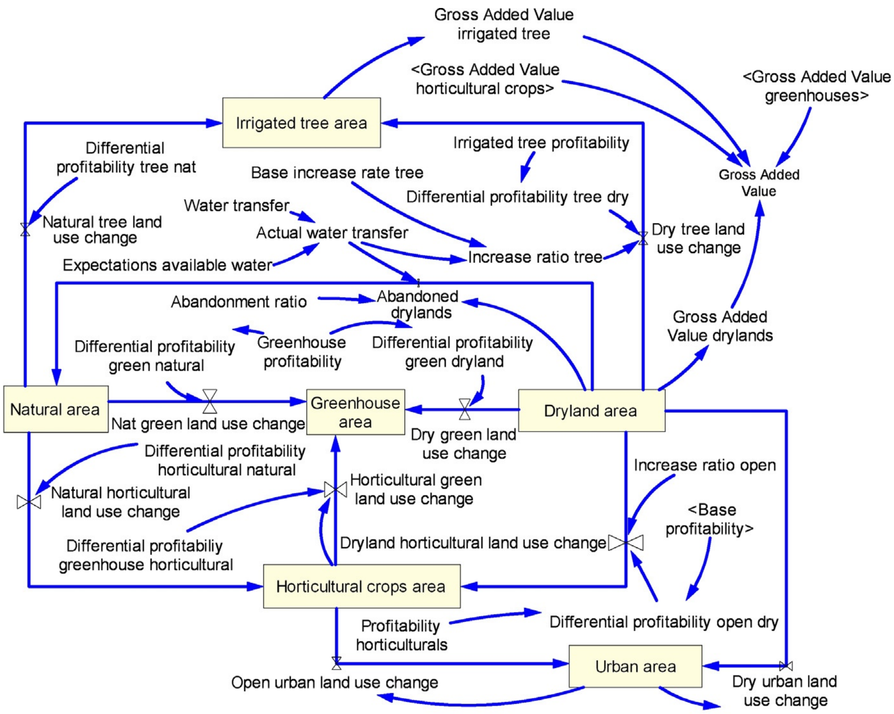

FIGURE 11.5

Simplified diagram of the land use model sector.

11.2.2 Simulation results

The model simulation shows a good degree of agreement with observed values for available data series. The model tracks the noticeable increase in irrigated lands driven by two factors: the water transfer for irrigation, which started in 1979, and the higher profitability of such agriculture compared with drylands, especially in the case of greenhouses (Figure 11.6).

This has generated an increased input of nutrients from diffuse sources into the Mar Menor (Figure 11.7). The estimated load of nutrients shows strong fluctuations due to the high variability in rainfall and the occurrence of flash-flood events, when large amounts of nutrients and materials from the watershed are flushed out and enter the lagoon. This is expected, since the hydrological regime in this arid Mediterranean area shows extreme differences in water discharges. Flash-flood events play a key role for the water, nutrients, sediment, and pollution flows entering the lagoon. For example, it has been demonstrated that dissolved organic pollutant concentrations in the Albujon, the main watercourse, during a flash flood are several orders of magnitude higher than in regular periods and that heavy rainfalls account for more than 70% of total input of many pesticides through the Albujon watercourse (Moreno-González et al., 2013). In watersheds with intensive agriculture in arid environments, big rainfall events and flash floods can mobilize the nutrients accumulated during several months or years. The Mar Menor wetlands, located along the lagoon shore, provide important ecosystem services in case of flash-flood events, with the storing and slowing down of floodwaters, allowing the nutrient removal of overland flow waters, and acting as sediment traps. Nutrient loads, particularly during flash floods, are linked to one of the most important problems for the bathing

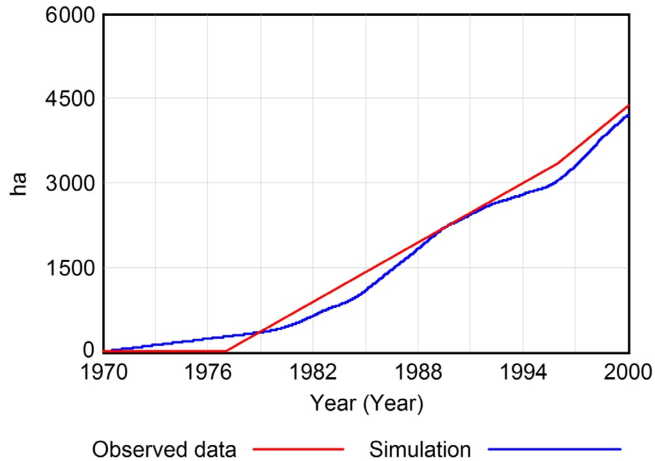

FIGURE 11.6

Area occupied by greenhouses in Mar Menor watershed. Observed and simulated values.

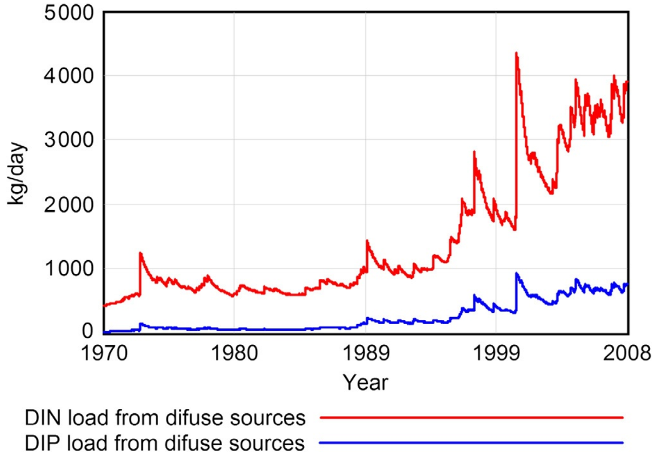

FIGURE 11.7

Simulated pattern of daily DIN (dissolved inorganic nitrogen) and DIP (dissolved inorganic phosphorus) input (kg/d) from diffuse sources using a 365-day moving average period.

quality and the socio-tourist uses: the jellyfish outbreaks (described later). Although not explicitly considered in the model, the sediment trapping is also linked to the ecosystem service of bathing quality. Mar Menor wetlands contribute to reduced water turbidity, mudding, and loss of lagoon depth (a problem already identified in the Mar Menor lagoon).

According to the simulation results, an average annual load of 900 ton/year of DIN (dissolved inorganic nitrogen) and around 200 ton/year of DIP (dissolved inorganic phosphorus) from diffuse sources can be estimated for recent years. These values are coherent with the scarce empirical data available on nitrogen content and water flows in the watershed (Lloret et al., 2005; Velasco et al., 2006; García Pintado et al., 2007; Álvarez-Rogel et al., 2009; Serrano and Sironi, 2009).

Population in the Mar Menor area has also shown a quick growth in recent decades due to tourism (Figure 11.8). The existence of treatment plants has led to a large increase in wastewater and the input of nutrients from urban sources, especially during summer (Figure 11.9), where the population of some locations increases more than 10 × (Moreno-González et al., 2013). After big rainfall events, overflow water from wastewater treatment plants enters the lagoon (Alvarez-Rogel et al., 2006), sometimes causing the temporary closing of the affected lagoon bathing areas (Conesa and Jiménez-Cárceles, 2007). The estimated average urban input is around 130 ton/year of DIN and 17 ton/year of DIP, which represents 12% and 8% respectively of all sources (point and diffuse). A large number of studies have shown that in watersheds dominated by intensive agriculture the main contribution to water pollution of rivers, lagoons, and coastal waters comes from diffuse sources and, more

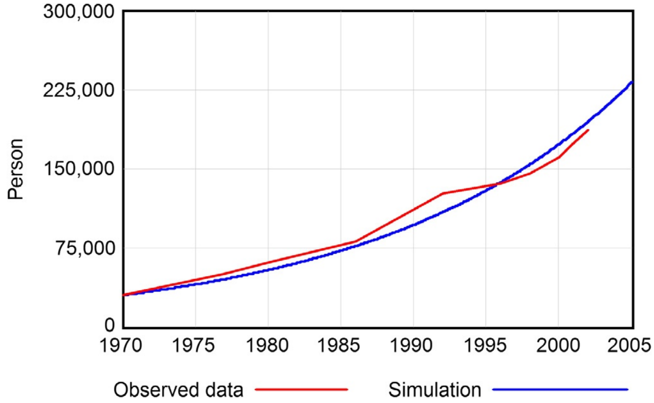

FIGURE 11.8

Peak summer population around Mar Menor. Observed and simulated values.

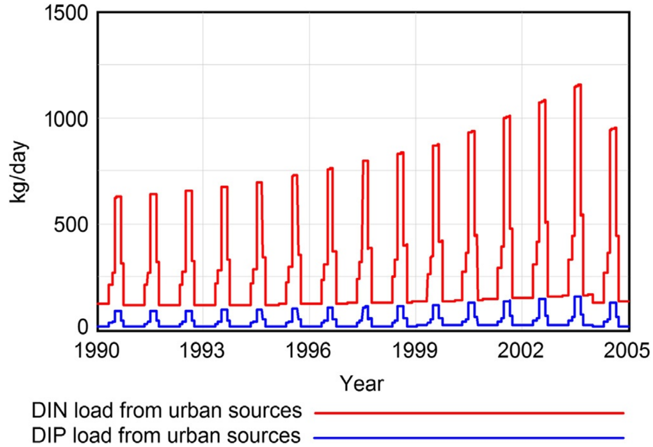

FIGURE 11.9

Simulated DIN (dissolved inorganic nitrogen) and DIP (dissolved inorganic phosphorus) input from urban sources.

specifically, from agriculture (Jordan et al., 1997; Kronvang et al., 1999; Meissner et al., 2002; Lacroix et al., 2005). As shown, this is also the case for the Mar Menor watershed, one of the most important intensive agricultural areas in Europe (Moreno-González et al., 2013). Has this increase in nutrients inflow affected the lagoon ecosystem? The next section considers this question.

11.3 EFFECTS OF HYDROLOGICAL CHANGES ON THE MAR MENOR LAGOON: JELLYFISH OUTBREAKS

The increased inflows into the Mar Menor lagoon have modified its structure and dynamics. An increase in nitrogen content of the water column has been observed throughout the last decades (Perez Ruzafa et al., 2002; Lloret et al., 2005; Velasco et al., 2006), which in one decade shifted to values 10× higher (Perez Ruzafa et al., 2002). The Albujon watercourse is the main source for nitrate in the lagoon waters. There are clearly defined spatial gradients of water column transparency and nutrient concentrations in the lagoon, resulting as a consequence of the inputs from the Albujon watercourse (Lloret et al., 2005).

The increased nitrogen content has favored other changes in the lagoon and its biological assemblages. One of the main changes is the existence of jellyfish outbreaks during summer. In 1974 the opening of the Estacio channel caused strong hydrographical changes, with a higher water exchange with the Mediterranean, a salinity decrease, and strong alterations in the biota. In the mid 1980s, two allochtonous species of jellyfish, *Rhyzostoma pulmo* and *Cotylorhyza tuberculata*, entered into the Mar Menor. The resulting moderated lagoon temperature and salinity ranges allowed these species to complete their biological cycle inside the lagoon (Pérez Ruzafa and Aragón, 2003). Scyphomedusae species were recorded for the first time around 1990. The summer proliferation of the two allochtonous species of jellyfish started during mid 1990s, in response to the increased nutrient inflow (Perez Ruzafa et al., 2002; Lloret et al., 2005, 2008; Conesa and Jiménez-Cárceles, 2007). Both species attained large populations, especially in the case of *C. tuberculata*, which in 1997 reached around 46 million individuals in summer (Perez Ruzafa et al., 2002). These large jellyfish populations affect nutrient cycles and other lagoon compartments, since outbreaks of jellyfish may reduce fish biomass (Purcell et al., 2007).

Due to their negative effects on the bathing quality and therefore on the tourist value of the Mar Menor lagoon, the Fish General Directorate of Region de Murcia took some measures to try to control the summer jellyfish blooms, including the annual catch of jellyfish during summer by means of special boats. Data on annual jellyfish catches and fishing effort provided by the regional authorities, in combination with data of total jellyfish population from direct census available for some years, allowed the estimation of the maximum summer population of jellyfish between 1988 (the starting date for the allochtonous species) and 2012 (Figure 11.10). There are noticeable interannual changes of the summer population of jellyfish, which peaked in 2000 and then began to decline, with no outbreaks detected in year 2005 (Prieto et al., 2010). In the following years the jellyfish population peaks remained at very low values with the exception of year 2006, when a small recovery was observed (Dolores et al., 2009). However, in the last several years the summer jellyfish populations have increased again. The reasons for such shifts are still under research, although water temperature seems to play a key role (Prieto et al., 2010; Astorga et al., 2012). For example, the low values of jellyfish population in 2005 and 2006 have been related to very low temperatures in 2004–2005 winter, which

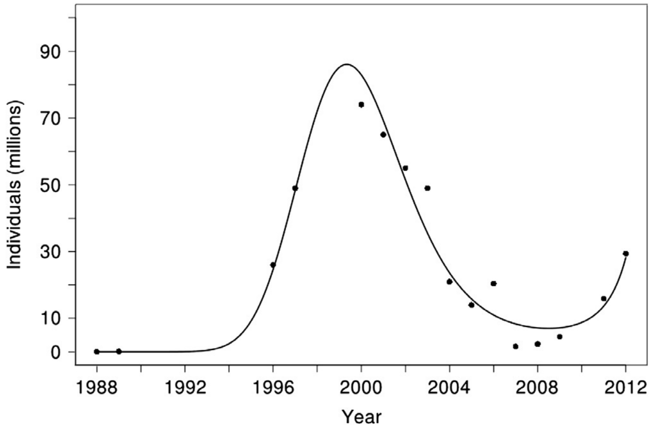

FIGURE 11.10

Estimated summer population of jellyfish in Mar Menor (dots) and Poisson adjustment (92.28% explained deviance, $p < 0.001$). Maximum summer populations were estimated from data on jellyfish catches, fishing effort, and total jellyfish population for available years provided by the Fish General Directorate of Murcia.

decreased polyp numbers in 2005, resulting in low jellyfish biomass also affecting jellyfish population in 2006 (Ruiz et al., 2012).

The increased nutrient inflows have caused other changes in the lagoon dynamics. The dominance shifts from the phanerogam *Cymodocea nodosa* to the macroalgae *Caulerpa prolifera* (Perez Ruzafa et al., 2002; Lloret et al., 2005; Conesa and Jiménez-Cárceles, 2007) are mainly explained by the inputs from the Albujon watercourse and associated changes in the water column parameters (Lloret et al., 2005). The observed changes in the macrophyte assemblages have other consequences for lagoon dynamics, such as the accumulation of organic matter under the meadows of *Caulerpa prolifera*; the subsequent appearance of anoxic conditions in some areas; and the decrease in populations of some commercial fish, mainly sparidae and mugilidae, which are negatively affected by the spread of the macroalga (Lloret et al., 2005).

11.4 ASSESSMENT OF ECOSYSTEM SERVICES: NUTRIENT REMOVAL

11.4.1 Role of wetlands and effects of measures to reduce nutrient inputs into the lagoon

At present, the main watercourse, the Albujon, is disconnected from the Marina del Carmoli wetland due to channeling works and, therefore, its full nutrient load enters the lagoon without benefiting from any removal function from the wetland.

However, overall, Mar Menor wetlands do intercept other ephemeral watercourses and, above all, they play a key role during big storms, reducing the nutrient load of flash-flood events. The watershed dynamic model has been used to estimate the proportion of nutrients being removed by the wetlands. According to simulation results, at present wetlands remove around 14% of total nutrients from diffuse sources.

As shown in previous sections, there is a clear need to achieve a more substantial reduction in the total load of nutrients reaching the lagoon. Moreover, according to model simulation results, the trend to an increase in the nutrient inflows would be maintained in the near future under a business-as-usual scenario characterized by urban and tourist development. Assuming that the overall performance of wastewater treatment plants is maintained, simulation results point to an increase in the input of nutrients from urban sources, especially during summer months. The need for a reduction in nutrient inputs is also a compulsory request by present European legislation: The Mar Menor watershed is designated as a Vulnerable Area according to the Nitrates Directive (91/676 ECC), the lagoon is designated as a Sensible Area according to the Urban Wastewater Directive (91/271 ECC), and the Mar Menor water should achieve and maintain good ecological status according to the Water Framework Directive (2000/60 EC).

The water body carried out a project for managing part of the agricultural drainage, consisting of several hydraulic facilities already built up, but not yet initiated, to collect part of the agricultural drainage coming from irrigated land. This drainage water will be collected and pumped to a desalination plant, after which it would be reused for irrigation. We have applied the dynamic model to simulate and assess the effectiveness of this management option to remove part of the nutrients reaching the lagoon and compare it with other management measures, particularly wetlands restoration. After implementing the measure of agricultural drainage management in the watershed dynamic model, simulation results suggest that this option might reduce the total nutrient inputs into the lagoon by around 10%.

The wetland restoration measure is based on (i) the restoration of part of the Marina del Carmoli wetland area lost due to land-use changes and (ii) the reconnection of the water flow of the Albujon watercourse to this restored area of the Marina del Carmoli wetland. According to model simulations, this measure would achieve around a 40% reduction in the amount of nutrients contributed by the Albujon watercourse. When all flows are considered, this management option still represents a noticeable additional reduction of around 20% in total nutrients input into the lagoon when compared with the base trend of urban and tourist development, doubling the reduction achieved by the management of agricultural drainage option (Figure 11.11).

11.4.2 Cost-effectiveness analysis

A cost-effectiveness analysis (CEA) has been applied to assess and compare the relative effectiveness of these two management measures (Martinez Paz et al., 2007). CEA is very useful to assess and select the management measures achieving the desired environmental goals at the lowest cost, which constitutes an important input for

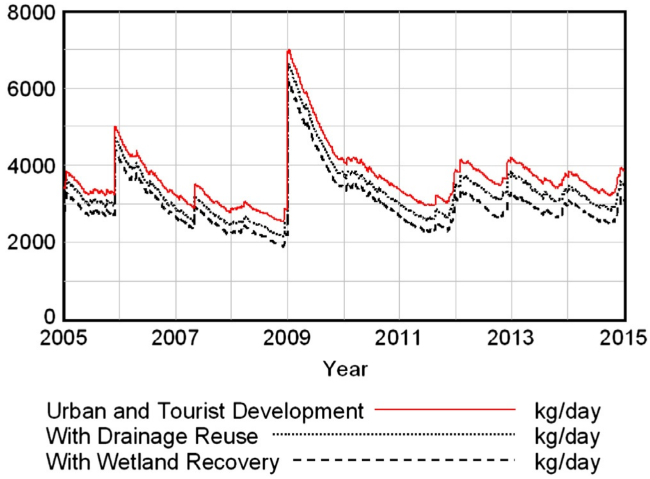

FIGURE 11.11

Simulated pattern of daily load of DIN (dissolved inorganic nitrogen) into the Mar Menor lagoon from surface water using a 365-day moving average period under the measures of management of agricultural drainage and restoration of wetlands, as compared with the base trend of urban and tourist development.

the decision process. CEA has been applied in other studies on the performance and efficiency of using wetlands versus conventional measures to treat pollution processes (Schou et al., 2000; Kampas et al., 2002; Zanou et al., 2003; Lacroix et al., 2005). Figure 11.12 shows the methodological steps of the CEA analysis in the Mar Menor area.

Several factors were quantified to determine the capacity and overall performance of the two options, including the average flows in the drainage channels, the effect of overland flow on the nutrient removal efficiency of the wetland during flash-flood events and the maximum capacity of the hydraulic facilities of the drainage management option, including the drainage channels, pumping station, and desalination plant. The maximum capacity of the drainage management system fits the requirements under normal conditions but cannot manage floods. A 15-year time frame has been considered to calculate the CEA. All items have been valued according to market prices. The land to be purchased has been valued as agricultural and the attributed land profit, the opportunity costs of its use as wetland, corresponds to the gross added value of intensive horticultural crops. The water price imputed to the drainage reuse option constitutes an income, since the desalinated water will be sold for irrigation. To compute the added financial flows, the net present cost (NPC) has been calculated using a low discount rate of 2%, according to the proposal by Almansa and Martínez-Paz (2011) for this type of project. The relative efficiency of the two management options has been compared by means

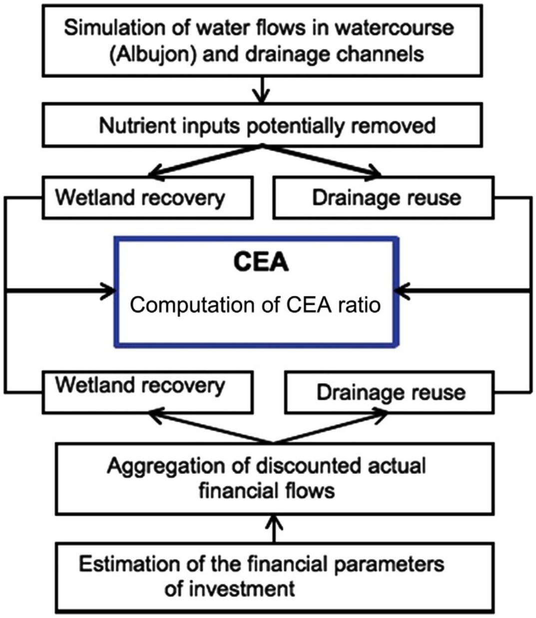

FIGURE 11.12

Basic methodological steps of the CEA applied in the Mar Menor site.

of the cost-effectiveness ratio (CER, Zanou et al., 2003). All costs refer to euros in real terms (2007). The cost-effectiveness ratio of each option is determined as the quotient between the amount of nutrient removed by each option during the study period and the aggregated costs during such period.

Results indicate that wetlands restoration is more cost-effective than the agricultural drainage management, the solution adopted by the water body, since its unitary costs in terms of euros per kg nutrient being removed are around half of that corresponding to the drainage management measure. Each 100 € invested in the drainage reuse system would remove 7.6 kg of DIN and 2.7 kg of DIP, whereas the removed nutrients would be double under the wetland restoration option. In the case of nitrogen, CER values are around 13 and 6.5 €/kg for agricultural drainage management and wetlands restoration measures respectively. This constitutes an important insight for the water body, which had not previously considered the option of wetlands restoration and never assessed the efficiency of the management of agricultural drainage, despite the fact that hydraulic facilities were already built up.

These results agree with other studies (Gren et al., 1997; Turner et al., 1999; Gustafson et al., 2000; Zanou et al., 2003; Lacroix et al., 2005), showing that the construction and especially the restoration of wetlands is a highly cost-effective option to reduce diffuse pollution in agricultural watersheds. Moreover, as explained above, the restoration of wetlands also achieves the highest nutrient removal in absolute terms.

Lacroix et al. (2005) also point out that wetlands restoration, an option usually considered as being very expensive, is frequently more cost-effective than other strategies such as subsidies to reduce the amount of fertilizers in agriculture.

11.4.3 Economic valuation of the ecosystem service of nutrients removal of Mar Menor wetlands

Despite the considerable uncertainties in the assessment of ecosystem services (Johnson et al., 2012), and the difficulties inherent to a proper estimation of the total value of such services, the information provided, even by incomplete or partial valuations, may become a useful tool to build social support for conservation policies and to assess relevant services for different users and management decisions (Martin-López et al., 2014; Nemec and Raudsepp-Hearne, 2013), in particular for local and regional planning (Maes et al., 2012).

The combined use of the watershed dynamic model and CEA analysis has allowed a first quantitative economic assessment of the ecosystem service of nutrient removal in the case of Mar Menor wetlands. To this aim, we have taken into account the estimation of nitrogen being removed by wetlands under present conditions according to model simulation results, which amounts to an annual average of around 193 ton/year of DIN for the recent period. If present wetlands are lost and this service should be supplied by other processes, particularly with the management of agricultural drainage, the CEA analysis results shown earlier indicate that the value would be around 2,509,000 €/year, which provides a value of around 7169 €/year per ha of active wetland (salt marsh + reed bed) for this specific ecosystem service.

In the same way, it can be estimated the avoided costs achieved by the restoration of wetlands. If the goal of additional 20% nutrients reduction achieved by this measure should be obtained by the management of agricultural drainage, the estimated additional required budget should be around 2,000,000 €/year, taking into account the difference in the unitary costs (CER) of both considered options.

We have also carried out a preliminary assessment of some other avoided costs associated with the ecosystem service of nutrients removal of Mar Menor wetlands. As described earlier, one important effect of the increased inputs into the lagoon is the jellyfish outbreaks. The high population densities take place during summer months, causing important disturbances for bathing and other tourist activities, the main socioeconomic activity of the Mar Menor area. As in many other coastal areas, high densities of jellyfish are detrimental to tourist appeal (Purcell et al., 2007). In response to the high jellyfish density values of 1996 and 1997, the regional authorities started the following programs during summer months as measures to reduce the jellyfish population in the lagoon: (i) the installation of temporary devices (meshes) to isolate bathing areas from jellyfish and (ii) catching jellyfish using special boats. According to data provided by the Directorate of Fish and Aquaculture, an annual budget varying between 200,000 and 400,000 €/year has been appropriated for the measure of jellyfish catch. For years with an average summer jellyfish density of around 3–4 individuals per 100 m^3 in the lagoon water, with total summer jellyfish catch varying between 1000 and 3000 ton (fresh weight), an average estimated cost of around 93 €/Kg N being removed from the lagoon by the jellyfish catches. On the basis of such value, the avoided cost of the nutrient removal service of Mar Menor wetlands in relation to jellyfish outbreaks can be estimated. As indicated earlier, the hypothetical loss of present wetlands would increase the annual input of DIN into the lagoon by around 193 ton/year. Taking into account the relationship between such inputs and the jellyfish population, it can be estimated that to counteract the expected additional jellyfish increase would require increasing the annual economic budget allocated to jellyfish catch by around 50–60%.

It should be noted that all these figures are not intended to represent the actual value of Mar Menor wetlands since only some aspects of their ecosystem services are considered. Moreover, the existence of dependencies between results from the valuation of ecosystem services and the methodological approaches that are applied (Martin-López et al., 2014) should be taken into account. However, this exercise constitutes a minimum estimate of the economic value of some of the avoided costs generated by such ecosystem service.

In synthesis, the use of Mar Menor wetlands seems to be more effective and economically efficient than other measures as the management of agricultural drainage to achieve the required reduction in nutrient flows from the watershed. Therefore, the enhancement of such ecosystem service should become an important goal under an integrated management and policy in the Mar Menor area. In the next section we present the effects of hydrological and nutrient changes in another important ecosystem service of the Mar Menor lagoon and associated wetlands: biodiversity conservation.

11.5 ASSESSMENT OF ECOSYSTEM SERVICES: BIODIVERSITY CONSERVATION

Quantitative studies carried out at European scale (Maes et al., 2012) have corroborated the positive relationships between biodiversity and a variety of ecosystem services. Such studies provide evidence on that a good conservation status usually means richer biodiversity and higher levels of ecosystem services. Therefore, it is important to assess the trends in the conservation status of biodiversity, particularly in ecosystems under pressure as the Mar Menor lagoon and associated wetlands. In this section we assess the recent trends in some components of their biodiversity: the aquatic bird assemblages and the habitats of Mar Menor wetlands.

11.5.1 Changes in aquatic bird populations in relation to nutrient inputs and related trophic variables: Indicator species and guilds

Aquatic birds are a key component among the ecological values on which the various protection designations of the Mar Menor are based. Since the inclusion of the lagoon and its associated wetlands in the Ramsar List in 1994 (Robledano, 1998), waterbird numbers have been the most popular and affordable criterion for assessing the conservation value of this ecological complex. In Murcia, the Mar Menor lagoon is the site with the longest dataset of wintering waterbirds and the one with fewest gaps (Robledano et al., 2011). One decade after its listing under the Ramsar agreement, the use of these monitoring data started to shift from a mere accounting of numerical trends to the search of indicator responses of bird metrics to recorded or modeled series of environmental data (Martínez-Fernández et al., 2005; Robledano and Farinós, 2010; Robledano et al., 2011). This has allowed the creation of a database of waterbird censuses coupled with environmental variables related to the trophic status of the lagoon, which now has been updated.

11.5.1.1 Waterbird data and environmental variables

Waterbird data have been obtained from the January censuses carried out between 1972 and 2013 in the framework of the International Waterbird Census (IWC), coordinated in Murcia Region by ANSE (Hernández and Fernández-Caro, 2008, unpublished data). The analysis is restricted to five species belonging to the families *Podicipedidae* (2), *Phalacrocoracidae* (1), *Anatidae* (1), and *Rallidae* (1), the criteria being to select only diving or dabbling waterbirds able to exploit the water column and benthos at depths of 1 m or more (i.e., strictly littoral dabbling species, wading birds, and shorebirds were excluded). The five target species (Great Cormorant *Phalacrocorax carbo*, Black-necked Grebe *Podiceps nigricollis*, Great Crested Grebe *Podiceps cristatus*, Red-breasted Merganser *Mergus serrator,* and Common Coot *Fulica atra*) are also the most abundant wintering waterbirds and those more closely tied to using the lagoon as a feeding and roosting area, thus representing the bulk of the biomass of top-level avian consumers. All species are zoophagous (fish or invertebrate feeders) with the exception of Coot (mainly phytophagous). Among the animal consumers, all are mainly piscivorous, although including varying proportions of other animal prey in their diet. With the exception of the Great Cormorant, which can make regular flights to feed at varying distances around the lagoon, all the target species spend all or most of the winter time in open water within the lagoon.

Regarding environmental variables, the estimated annual inflow of nitrogen and summer jellyfish population presented earlier is used, as well as official data on fish landings declared by the two main fishing harbors of the lagoon as a proxy of fish production, focusing on the main potential prey species for piscivorous waterbirds (*Engraulis* sp., *Atherina* sp.). Further details on data sources and their processing can be found in the previous sections and in the cited publications. An overall graphical summary of the waterbird and environmental data is shown in Figure 11.13.

11.5.1.2 Statistical analysis

Generalized linear models (GLMs) with a Gaussian link were used to model the relationship between the biomass of waterbirds (computed as the product of census results by mean species' weight values from bibliography) and the environmental variables. Models were built for individual species and higher taxonomic or functional aggregations (families, guilds). As functional aggregations, we refer to groups of species with similar feeding ecology and distribution, of which we considered

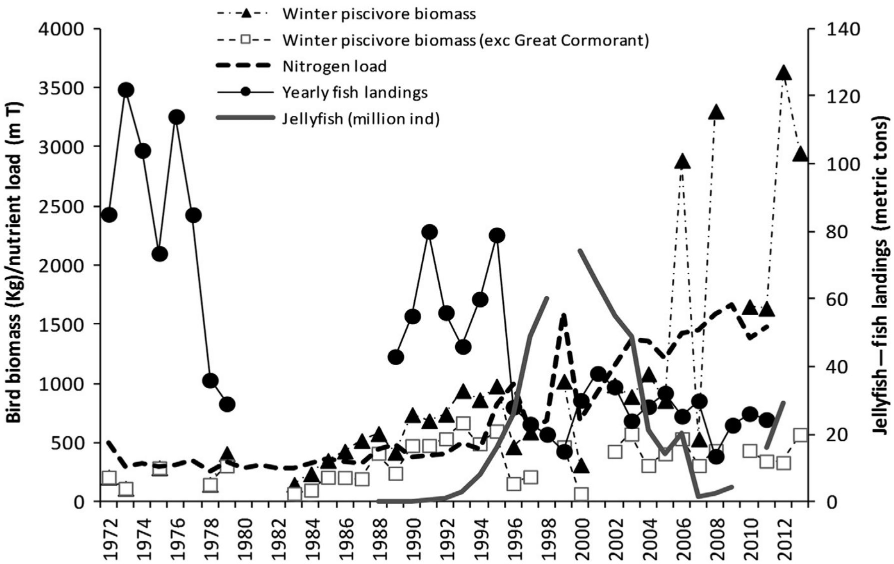

FIGURE 11.13

Changes in the environmental variables tested in regression models in relation to the variation of the main components (piscivores) of the waterbird assemblage of the Mar Menor lagoon.

two: (i) all piscivores and, nested within it, (ii) lagoon-restricted piscivores (excluding Great Cormorant, due to its mobile feeding strategy). Regression models were fitted in the freely distributed statistical software R. Model performance was evaluated using Akaike's information criterion (AIC), which seeks to optimize the trade-off between the explanatory ability of the model and its complexity, measured by the number of fitted parameters. The amount of variability explained by each model was assessed by means of its reduction in deviance with respect to the null (intercept-only) model (Saunders et al., 2013). Models were fitted sequentially by first selecting the best performing model with one of the two variables related to the trophic status of the lagoon (nutrients or fish), and later including jellyfish as potential modifiers of such status, as well as the interactions among the first and second variables. Quadratic terms were also tested for the trophic-related variables, in search of a best fit of the dependent variable. Nutrients and fish were included separately in the models due to their redundancy (Spearman's Rho $=-0.704$, $p<0.0001$). Nutrient load was tested with a lag of 2 years, considering the best response models of previous studies (Robledano et al., 2011), and jellyfish with 1- or 2-year lags.

11.5.1.3 Results and discussion

All species except the Red-breasted Merganser *Mergus serrator* showed a positive response to nutrient inputs and a negative one to fish catches (the latter was not tested for the Coot for its obvious lack of biological meaning). Models including the 2-year lagged estimate of annual nitrogen load usually explained more variation (or percent deviance) than those including the fish catch (Table 11.1). Although the Red-breasted Merganser can be considered a specialist piscivore, its relationship with the availability of its main prey (indicated by fish landings) was not statistically significant, while it showed a significant negative response to nutrient loading. The only model including an interaction is that of the Coot with nutrients and jellyfish (both with a 2-year lag). The net positive effect is not surprising since the control exerted by jellyfish on eutrophication is expected to affect mainly the growth of phytoplankton, but not necessarily other food sources preferred by phytophagous birds (e.g., filamentous algae).

The general picture, for most species and higher taxonomic or functional groupings, is an overall increase in biomass positively correlated with the increase in nutrient inputs. On the contrary, both waterbird biomass and nutrient inputs show a negative relationship with fish resources (Table 11.1). This global picture does not disagree with previous interpretations of waterbird responses, since the species contributing to the global trend vary along the period of study (Figure 11.13), allowing recognition of six temporal phases characterized by specific responses to the status of the lagoon (Figure 11.14). All except one of these phases have been described previously (Robledano et al., 2011) and can be summarized as:

1972–1979: Characterized by low values of nutrient inputs and high fishing yields, starting to decrease at the end of the period, probably due to overfishing; the waterbird assemblage is dominated by the specialist Red-breasted Merganser in all years except 1979.

1983–1987: Nutrient inputs are still low but steadily increasing, and a first positive response of generalist piscivores (mainly Great Cormorant) is detected.

1988–1995: Characterized by a gradual increase in nutrient inputs and by a growing numerical contribution of *Podicipedidae* with respect to Red-breasted Merganser and Great Cormorant.

1996–1998: A period characterized by low fishing yields, higher nutrient loads, and an incipient development of jellyfish, coincident with a diversification of the waterbird assemblage (the dominance of *Podicipedidae* decreases and Coot starts to be recorded in low numbers).

1999–2006: Shows additional increases in nutrient inputs and a higher abundance of jellyfish, which can exert some control on eutrophication, keeping a more fluctuant but relatively diverse waterbird assemblage (with lack of a clear dominance among species). The greatest abundance of Coot occurs during this phase, rising to a dominant position in 2006 and decreasing later.

2007–2013: The more recent phase, representing a return to a less diverse waterbird assemblage, dominated by two generalist piscivores: Black-necked Grebe and Cormorant. Red-breasted Merganser and Coot become scarce or almost disappear from the lagoon. If Cormorant is excluded, a stabilization of waterbird numbers around somewhat lower values (compared with precedent maxima) can also be stated.

Table 11.1 Selected GLMs Explaining Numbers and Biomass of Individual Waterbird Species and Higher Taxonomic or Functional Aggregations

Dependent Variable	Estimate	*P*	AIC	Explained Deviance (%)
Biomass of *Phalacrocorax carbo* (Null)			527.13	
Biomass of *Phalacrocorax carbo* ~ Nload.2yr	1.40	***	500.74	58.82
Biomass of *Phalacrocorax carbo* ~ Fish_land	−13.19	*	382.44	23.84
Biomass of *Podiceps nigricollis* (Null)			397.75	
Biomass of *Podiceps nigricollis* ~ Nload.2yr	0.16	***	366.63	63.34
Biomass of *Podiceps nigricollis* ~ Fish_land	−1.45	*	294.08	19.34
Biomass of *Podiceps cristatus* (Null)			391.98	
Biomass of *Podiceps cristatus* ~ Nload.2yr	0.07	*	388.88	14.32
Biomass of *Podiceps cristatus* ~ Fish_land	−0.20	NS		
Biomass of *Mergus serrator* (Null)			397.37	
Biomass of *Mergus serrator* ~ Nload.2yr	−0.11	***	386.99	31.27
Biomass of *Mergus serrator* ~ Fish_land	1.10	NS		
Biomass of *Fulica atra* (Null)			374.34	
Biomass of *Fulica atra* ~ Nload.2yr	0.17	***	361.67	38.67
Biomass of *Fulica atra* ~ Jfish.2yr	3.54	***	187.76	43.86
Biomass of *Fulica atra* ~ Nload.2yr*Jfish_2			174.52	82.22
Nload_2	−0.15	0.08		
Jfish_2	−6.73	*		
Nload.2yr*Jfish	0.01	**		
Biomass of *Podicipedidae* (Null)			434.12	
Biomass of *Podicipedidae* ~ Nload.2yr	0.23	***	417.36	43.36
Biomass of *Podicipedidae* ~ Fish_land	−1.65	NS		
Biomass of Piscivores exc *Phalacrocorax carbo* (Null)			434.51	
Biomass of Piscivores exc *Phalacrocorax carbo* ~ Nload.2yr	0.11	0.05	432.56	11.29
Biomass of Piscivores exc *Phalacrocorax carbo* ~ Fish_land	−0.55	NS	330.15	
Biomass of Total Piscivores (Null)			547.82	
Biomass of Total Piscivores~ Nload.2yr	1.49	***	521.89	57.10
Biomass of Total Piscivores~ Fish_land	−12.77	*	403.18	19.07

*Significance. ***: p<0.001; **: p<0.01; *: p<0.05; NS: No Significant*

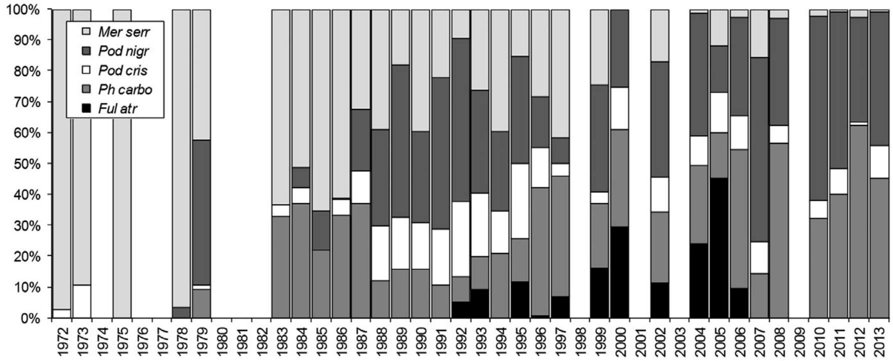

FIGURE 11.14

Percent contribution of the five species studied to the total abundance of waterbirds wintering in the Mar Menor Lagoon (January censuses).

The last phase is characterized by high nutrient inputs and similarly high numbers of the two dominant species: Great Cormorant and Black-necked Grebe. Also evident is the dramatic decrease that confirms the long-term decline of Red-breasted Merganser, once the most characteristic species of the lagoon. The two dominant species do not differ markedly in numbers (mean $\pm$ SE for the period 2007–2013, Cormorant: 861.3 ± 217.8, Black-necked Grebe: 814.5 ± 92.4), but do so in terms of biomass (Cormorant: 1879.4 ± 475.2, Black-necked Grebe: 814.5 ± 92.4).

Regarding the whole period of analysis, the apparent positive correlation with nutrient inputs is not exclusive of piscivores, since the only phytophagous species (Coot) shows the same response (Table 11.1). Interestingly, Coot biomass is also positively associated with the abundance of jellyfish in the lagoon (both thrive in the lagoon during the same period), which can be due to an independent response to a common underlying factor not captured by our environmental variables or to the interaction among them reflected by the models. Whatever the relationship between this short-term increase of herbivores and the blooms of jellyfish, the greater abundance of Coot could be related to a phase of high input of nutrients of urban origin (untreated sewage). This organic pollution has reached the lagoon through its main incoming watercourse (Albujón channel; García Pintado et al., 2007), and combined with the diffuse sources of nutrients of agricultural origin, may have favored the increase in wintering Coot, which are mainly restricted to the area of influence of the watercourse. After this phase, the increases in the two presently dominant species (Cormorant and Black-necked Grebe) would be explained by the persistent high loads of agricultural nitrogen, although the precise mechanisms by which such increased fertilization is transferred to waterbirds through the lagoon's trophic web are more difficult to establish. Clearly it is not a matter of increased fish resources, at least of the species accounted for by our variable.

In any case, correlative studies like that presented here do not necessarily demonstrate causal relationships (i.e., a direct effect of nutrient load on some waterbird food resource). Nor do the increase in bird abundances necessarily relate to local environmental factors. Recent studies have tried to discriminate which factors, operating at different geographical scales—from the site where the waterbirds are counted to the whole biogeographical population range—explain the change in waterbird numbers recorded by IWC and other monitoring schemes (Tománková et al., 2013). In our case, a statistical dependence on external demographic trends (increase in the western Mediterranean biogeographical population) had been established only for the Great Cormorant (Robledano et al., 2011). The ability of Cormorants to take advantage of new sources of food is one of the causes of its increase and could explain its trend without a more direct local trophic response. If the Great Cormorant is not included in the analysis, the total waterbird biomass during the last phase (2007–2013) is still higher than the initial values (405.9 ± 39.5 vs. 204.9 ± 50.2 kg in 1972–1975), but has decreased in relation to the peak values of the early 1990s (603.6 ± 56.4 kg, 1990–1995) and 2000s (617.7 ± 111.1 kg, 2000–2005).

On the other hand, the relationship of piscivorous waterbirds with fish resources might be obscured by the fact that we use a restricted set of fish catch statistics (two species). In the recent years there has been a recovery of commercial fish species not included in our dataset (e.g., *Sparus aurata*; García et al., 2001; Centro Regional de Estadística de Murcia, 2013), which could be exploited by large piscivores like the Great Cormorant. In fact, the contribution of the Mar Menor lagoon to the total number of wintering Cormorants in Murcia Region rose from ca. 30% to ca. 60% between 2003 and 2013. Since the regional population experienced also an overall increase during that period (Hernández and Fernández-Caro, 2008; ANSE, unpublished data), it is feasible that the trophic resources offered by the lagoon (or by the foraging range centered on it) have also increased. Certainly such an increase may be partially related to the increased fertilization of the lagoon, although a species as mobile as the Cormorant could also exploit other resources located within its daily flight range (e.g., fish stocked in irrigation infrastructures, open-sea aquaculture facilities, rivers, or reservoirs).

Different factors need to be invoked when explaining the recent numerical trend of the Black-necked Grebe, a species much more closely tied to the Mar Menor lagoon as feeding area and with a diet based on smaller prey. In this case the pathways through which the ongoing nutrient enrichment of the lagoon could be turned into increased food resources do not seem to flow through the fish compartment (at least the commercially exploited stock). The most plausible explanation for the recent numerical increase of Black-necked Grebe would be an enhanced supply of benthic macroinvertebrates and non-commercial fish thriving in *Caulerpa prolifera* algal beds, which are also thought to be responsible for the sequestration of large amounts of nutrients from the water column (Lloret and Marín, 2009). This is consistent with the fact that nutrient enrichment has not resulted yet in a hypereutrophic, phytoplankton-dominated status (even with jellyfish absent). It is also evident that much fewer food resources would be needed to sustain a population of Grebes than a equivalent one of Cormorants. Regarding the potential effect of external factors affecting waterbird populations at higher scales, the population of Black-necked

Grebes wintering in Murcia region is also increasing, which would suggest a common demographic response to higher-scale environmental drivers (e.g., hydrographic basin-level eutrophication or widespread artificial wetland creation). The Mar Menor population of Black-necked Grebes represents, on average, 60% of the whole regional population (2003–2013; Hernández and Fernández-Caro, 2008; ANSE, unpublished data), and this percentage has not changed markedly during the last decade. Even under a scenario of regional increase of its wintering population, it cannot be discarded that the capacity of the lagoon to host wintering grebes has been enhanced by internal nutrient loading.

11.5.1.4 Concluding remarks

As in previous studies, waterbirds provide some useful indication of changes in the trophic status of the Mar Menor lagoon. The replacement of dominant species and the resulting changes in the composition of the waterbird assemblage are interpretable on the basis of temporal phases characterized by specific syndromes of environmental pressures and impacts. To summarize the direction of change overriding such succession of waterbird assemblages, the biomass of avian consumers has increased over a 40-year period of variable but continuously increasing nutrient inputs, also characterized by a dramatic decline in fishing yields. *Mergus serrator*—the initially dominant and almost exclusive piscivore—has declined, while generalists like *Phalacrocorax carbo* and *Podiceps nigricollis* have continued to increase until becoming dominant in a simplified piscivore assemblage. While these two species could also be responding to more global environmental trends—e.g., recovery of previously endangered populations, background eutrophication at a regional or hydrographic basin scale—they doubtlessly portray a locally forced ecosystem status. The return to a more diverse assemblage of piscivorous waterbirds, characteristic of a less stressed situation, would indicate some recovery of local environmental quality. This intermediate assemblage, however, would still be far from the original community typical of oligotrophic and hypersaline waters. The species contributing to the diversification of the waterbird community during these intermediate stages (Great-crested Grebe, Coot) should be rather watched as early warnings of the disrupting effects of agricultural or urban eutrophication on the lagoon's ecology.

11.5.2 Effects of hydrological changes on habitats and vegetation dynamics

11.5.2.1 Habitat changes and their relationships with land use change in the watershed

Land use changes in the watershed and associated hydrological changes have caused a rise in water tables of aquifers and have increased the levels of groundwater, flooding periods, and soil water content in the wetlands (Alvarez-Rogel et al., 2007). We have studied the vegetation changes in the Marina del Carmoli wetland and to what extent such changes are caused by the increased inflow of water and nutrients from the watershed.

As all Mar Menor wetlands, the Marina del Carmolí wetland comprises salt steppe areas, salt marsh, reed beds, and a narrow sandy strip on the waterfront. Vegetation units are distributed according to the availability of water and its salinity. Salt steppes are located in areas with low water availability; reed beds are in areas with high water content and low salinity, whereas salt marsh occupies areas with intermediate water content and higher salinity. We studied the changes between 1984 and 2009 regarding the area occupied by the wetland and the major vegetation units and land cover classes: salt steppes, salt marsh, reed beds, crops, water bodies, bare soil, and infrastructures. Remote sensing techniques were applied based on Landsat TM and ETM+ satellite images using supervised classification (Carreño et al., 2008).

Results show that in 1984 Marina del Carmolí was mainly covered by salt steppe, which comprised an area of 243 ha, whereas in 2009 this habitat had lost more than a half of its original area. On the contrary, reed beds, practically absent in 1984, occupied an important area in 2009 (165 ha), after an important expansion process since 1995 (Figure 11.15). The relative changes between salt steppe, salt marsh, and reed beds can be explained by the interaction between soil moisture and conductivity gradients. The initial increase of water inflows from the basin resulted in increased soil moisture and higher salinity, which favored the expansion of salt marsh at the expense of salt steppe. At a later stage, around 1995, the increased water inputs reduced water salinity and allowed the expansion of reed beds. Figure 11.16 shows this pattern of change across time.

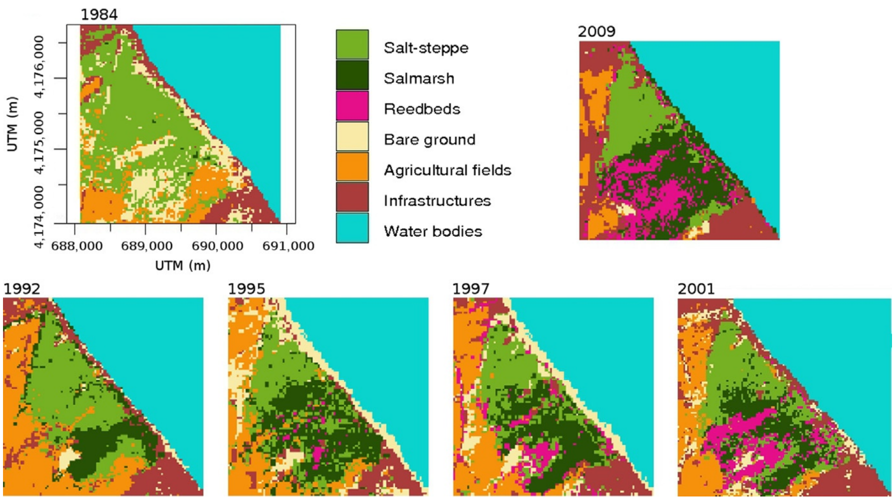

FIGURE 11.15

Maps of vegetation units and land cover of Marina del Carmolí wetland in 1984, 1992, 1995, 1997, 2001, and 2009, obtained by supervised classification of Landsat TM and ETM+ images.

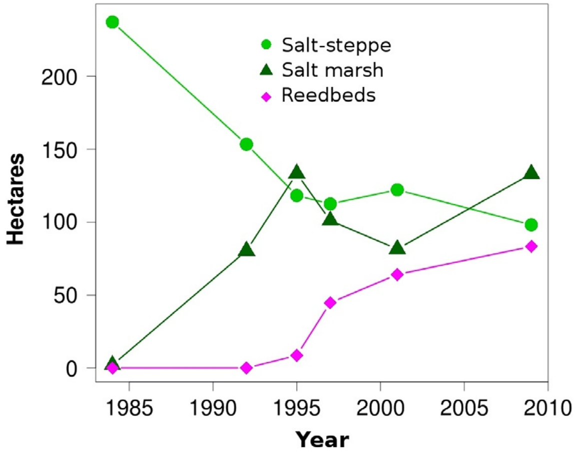

FIGURE 11.16

Area occupied by salt steppe, salt marsh, and reed beds in the Marina del Carmolí between years 1984 and 2009. Data obtained by supervised classification of Landsat TM and ETM + images.

Changes in land use are the primary factor explaining the described changes in the habitats of Mar Menor wetlands, as found in many other studies (Gustafson and Wang, 2002; Liu et al., 2004; Parra et al., 2005; Olhan et al., 2010). Figure 11.17 shows the close relationship between the expansion of irrigation in the Mar Menor basin and the active wetland area (salt marsh and reed beds) in the set of wetlands associated with the inner shore of the Mar Menor lagoon (Marina del Carmolí, Lo Poyo and Playa de la Hita). The regression model supports this close relationship, especially when considering a 5-year time lag (Figure 11.18, $R^2_{adj} = 0.945$, $p < 0.001$), a period that can be considered as the time required for the habitat to respond to the increased water inflows (Carreño et al., 2008; Esteve et al., 2008).

An important question arises regarding whether the changes in the structure of vegetation of the Mar Menor wetlands (Marina del Carmolí, Lo Poyo, and Playa de la Hita) have also modified the biodiversity values supporting the designation of these wetlands as protected sites. We answer this question in the next section.

11.5.2.2 Assessment of changes in biodiversity and conservation value of Mar Menor wetlands

Despite the growing research on biodiversity and conservation issues, there are still important knowledge gaps and lack of empirical evidence on the role of species loss or changes in species composition in maintaining ecosystem services (Mertz et al., 2007; Bastian, 2013). However, the protection status of species or communities represents an important consensus on the importance of biodiversity and therefore it may constitute the basis for assessing the changes in biodiversity and conservation

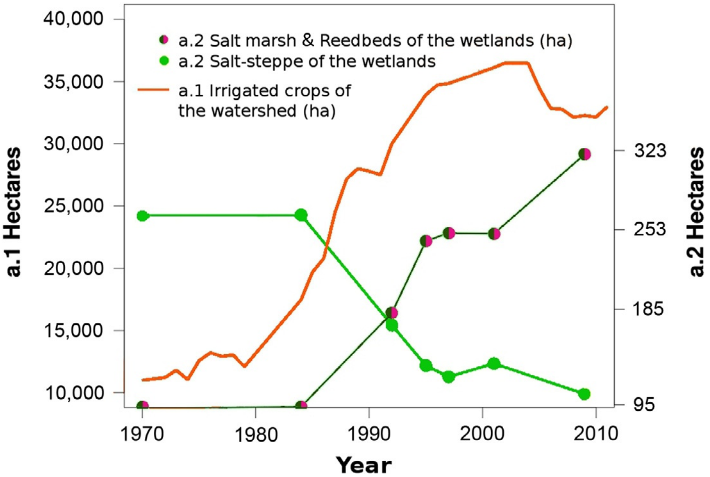

FIGURE 11.17

Axis a.1: Area of irrigated lands in the Mar Menor watershed; axis a.2: area occupied by salt steppe and area occupied by active wetland (salt marsh plus reed beds) in the wetlands associated with the inner shore of the Mar Menor lagoon (Marina del Carmolí, Lo Poyo and Playa de la Hita).

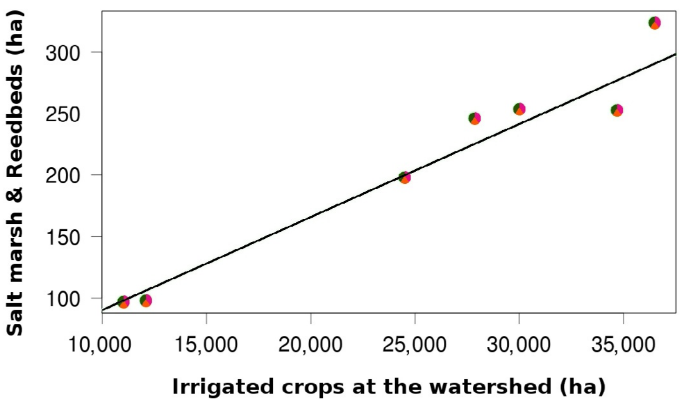

FIGURE 11.18

Regression model between the irrigated area of the Campo de Cartagena and the area occupied by salt marsh plus reed beds in the wetlands at the inner shore of the Mar Menor lagoon (Marina del Carmolí, Lo Poyo and Playa de la Hita) with a 5-year delay interval ($R^2_{adj}=0.945$, $p<0.001$).

value. Protected species lists, biological value indices, species population modeling, and scenarios development can help economic valuation and decision making (Mertz et al., 2007; Bastian, 2013). Here we assess the changes in biodiversity and conservation value of Mar Menor wetlands from the point of view of the protection status of habitat and species regarding the EU directives, particularly the Habitats directive.

Along with the Birds Directive, the Habitats Directive constitutes the basis for the Nature 2000 network of protected sites, designed to conserve European biodiversity, particularly important habitat and endangered species. In the Mar Menor wetlands, following the typology of the Habitats Directive, the salt steppe unit is 95% composed by habitat 1510, "Mediterranean salt steppes." The salt marsh unit mostly consists of habitat 1420, "Mediterranean and thermo-Atlantic halophilous scrubs." Finally, the reed beds unit is dominated by *Phragmites australis*, which is not included in the Habitats Directive. Salt steppe is considered of priority interest by the Habitat Directive; the salt marsh is of community interest; and the reed beds are not included in the Directive.

According to the data obtained with remote sensing described above (Figure 11.16), between 1984 and 2009 the area of salt steppe, of priority interest, has been reduced to less than the half; the area of salt marsh, of community interest, has doubled; and reed beds, without interest from the point of view of the Directive, has multiplied fivefold. The net loss of salt steppe is very important since it is the habitat of Mar Menor wetlands with the highest interest from the point of view of the Directive. Moreover, salt steppe is a rare habitat, comprising only 12,976 ha in Spain. Therefore, any reduction in this habitat constitutes a noticeable loss, particularly taking into account that the conservation status of this priority habitat in Murcia is much higher than the average in Spain, with 83% and 37% in good condition status respectively (Esteve and Calvo, 2000).

To quantify the relative change in biodiversity and conservation value of wetland vegetation from the point of view of the EU Habitats Directive, it has been proposed and applied an index as the weighted average of the area occupied by each vegetation type, assigning the values 0 (no interest), 1 (community interest), and 2 (priority interest) to the reed beds, salt marsh, and salt steppe communities, respectively. This index ranges between 0 (minimum value) and 2 (maximum value). As shown (Figure 11.19), the changes have resulted in an overall reduction of 48% in the biodiversity value and conservation interest of the vegetation from the perspective of the EU Habitats Directive. This is a worrying issue since the Marina del Carmoli wetland as been designated as SCI on the basis of this directive.

Moreover, changes in the steppe bird community of Marina del Carmoli has been described in response to the increased flows affecting this wetland (Robledano et al., 2010). A Birds Directive ((79/409 ECC) based index was proposed and applied. This index of conservation status takes into account the abundance index (IKA) of species and their inclusion in Annex I of the EU Bird's Directive. Results showed a marked decline in this index along the 1984–2008 period (Robledano et al., 2010). Since the Carmoli wetland has also been designated as a Bird SPA (Special Protection Area) under the Birds Directive, this loss in biodiversity and conservation value from the

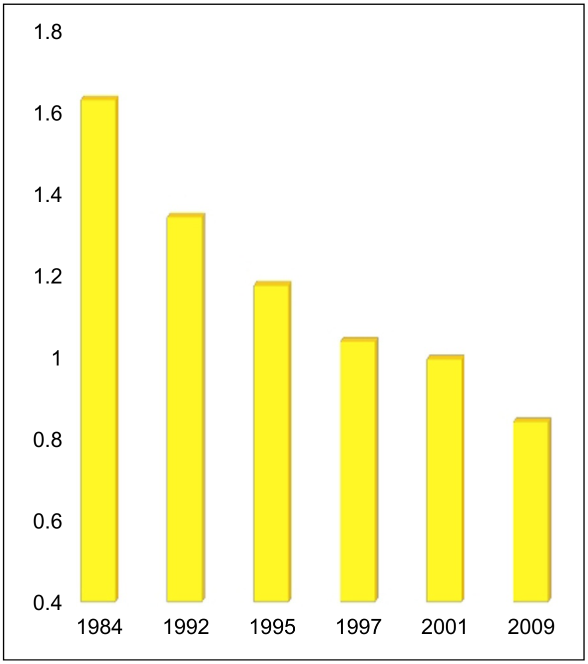

FIGURE 11.19

Evolution between 1984 and 2009 of the index that expresses the interest of vegetation at the wetlands of the inner shore of the Mar Menor lagoon (Marina del Carmolí, Playa de la Hita, and Lo Poyo) from the point of view of the EU Habitats Directive.

perspective of the EU Birds Directive is also of special concern. The effects of hydrological changes on other biological assemblages of wetlands not included in the EU directives, such as ground beetles (Pardo et al., 2008), are also coherent with the trends of change shown by habitats and the steppe-bird community.

Besides, these results also show that conventional protection and conservation strategies usually do not take into account the close dependency of wetlands on the dynamics and management outside the protected area and that this may interfere on the protection and conservation goals of protected wetlands.

11.6 TRADE-OFFS BETWEEN ECOSYSTEM SERVICES OF WETLANDS

The described changes in the Mar Menor lagoon and its biological assemblages (jellyfish outbreaks, aquatic birds) show the need for measures aiming at reducing the nutrient flows. According to results using the dynamic model and the cost-effective analysis, the enhanced use of wetlands to remove part of such flows is more effective that the management of agricultural drainages, which is the solution initially considered by the water body.

However, the increased water and nutrient flows have reduced the biodiversity and conservation value of wetlands, as derived from the species and biological groups supporting the designation of such wetlands as protected sites on the basis of the European Bird and Habitat directives, as the indexes applied to the habitats and to the steppe-birds community have shown. Therefore, additional increases of water flows into the wetlands, as required to increase the amount of nutrients being removed and reduce the ongoing eutrophication process in the lagoon, might cause further losses in the biodiversity and conservation value of wetlands. This points to a potential trade-off among ecosystem services of wetlands, particularly in arid environments, since they frequently present particular characteristics such as hypersaline conditions and low water flows (crypto-wetlands), characteristics that are both highly vulnerable to hydrological changes and also the basis for a specific biodiversity, which is considered rare in the European context.

The importance and magnitude of the indicated trade-off might be higher in the short future since (i) water and nutrients flows might increase, as shown by the simulation of the base trend scenario; (ii) there is a need to substantially reduce the nutrient flows into the lagoon, according to the Wastewater, Nitrates, and Water Framework directives; (iii) the management of wetlands for nutrients removal seems to be the most effective and economically efficient measure to achieve such goal; and (iv) increased water flows into the wetlands, as required to enhance the ecosystem service of nutrients removal and reduce the flows into the lagoon, would further threaten the biodiversity of Mar Menor wetlands and their conservation value according to the Birds and Habitat directives.

Some strategies and management options may help to solve such potential trade-offs. One such strategy is to spatially differentiate—to some extent—both ecosystem services of wetlands by allocating further enhancements of the nutrient removal function outside the boundaries of the protected site. This can be carried out by means of the restoration of lost areas of wetlands or the creation of new wetland areas at the expense of marginal crops. This is the case of the wetlands restoration measure proposed in Marina del Carmoli (Figure 11.20) to restore part of the original wetland area, presently occupied by marginal crops and to reconnect the Albujon watercourse, at present disconnected from the wetland, in order to manage and remove the nutrients transported by this important watercourse before entering the lagoon.

There is considerable evidence about important trade-offs between some ecosystem services, particularly between provisioning services (e.g., food production from agriculture) and regulating services such as soil control and water quality, trade-offs that should be taken into account in ecosystem management (Maes et al., 2012). This has also been pointed out in the case of wetlands (Maltby and Acreman, 2011), including potential trade-offs between nutrient removal and species richness of wetlands (Zedler and Kercher, 2005; Verhoeven et al., 2006; Maltby et al., 2013). However, as shown in the Mar Menor case, these trade-offs are particularly

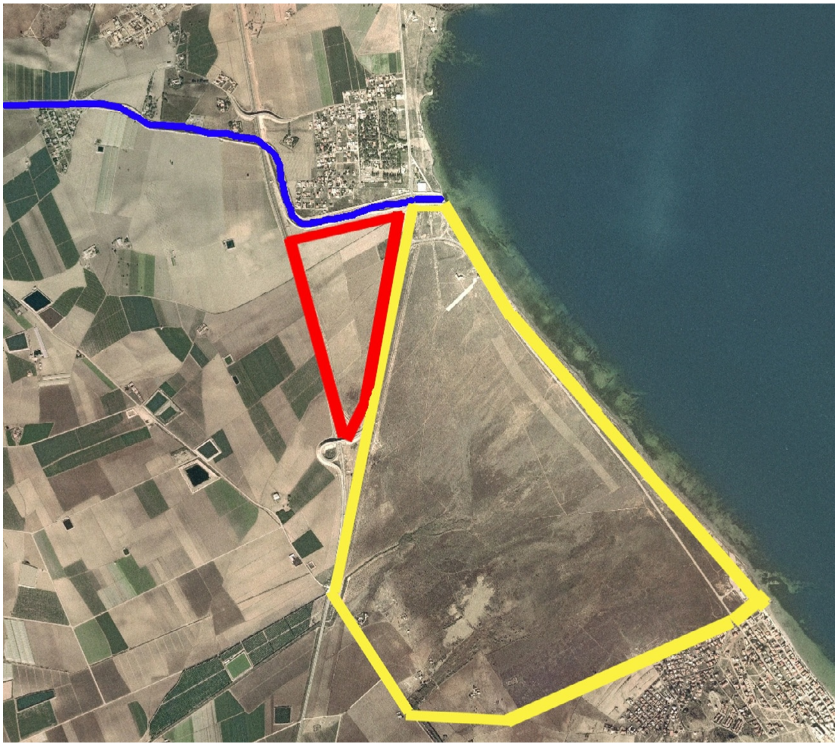

FIGURE 11.20

Area for proposed restoration of wetland area in Marina del Carmoli. Yellow: present protected site; red: crops where the original wetland area might be restored; blue: Albujon watercourse, at present disconnected from the wetland.

important in the case of wetlands in arid environments, where the enhancement of the nutrients removal service may alter their low water and high saline conditions and associated biodiversity values.

This highlights the need for integrated approaches under which synergistic relationships and potential trade-offs between ecosystem services can be properly considered for better informed decisions. Modeling how land use changes and management decisions impact multiple ecosystem services remains as an important challenge (Nemec and Raudsepp-Hearne, 2013). As a contribution to such a challenge, ongoing research on Mar Menor wetlands will focus on a full integration of biodiversity and conservation status indicators and the assessment and valuation of ecosystem services within the integrated dynamic modeling framework for the Mar Menor area.

11.7 CONCLUDING REMARKS

In the light of results shown in the Mar Menor case, we can go back to the questions regarding the role of models for the assessment of ecosystem services. Results shown in this work illustrate how the Mar Menor modeling framework has been useful to assess the ecosystem services of wetlands, taking into account the issues outlined in the introduction: the wetland-watershed relationships, the drivers of change (irrigation), the identification of less common but site-important services (effect on jellyfish outbreaks through the control of nutrient loadings into the Mar Menor lagoon), and the potential interactions among wetland ecosystem services (trade-offs between nutrients removal and biodiversity conservation).

Models help to estimate important factors for ecosystem services assessment that are not empirically available. In the Mar Menor case, the modeling framework has allowed a first valuation of the nutrient removal service of their coastal wetlands. Models are not only suitable for describing the structure and function of ecosystems but also to show how external drivers like land use affect them and their ecosystem services (Galic et al., 2012), to reveal trade-offs and to assess the change of ecosystem services along time under different scenarios to inform adaptive management (Maltby et al., 2013). For example, models may help to address some of the identified knowledge gaps, such as how much can we increase the nutrients inflow in a certain wetland without compromising its biodiversity values (Zedler and Kercher, 2005). Models can also help to provide evidence regarding the benefits of more integrated wetland ecosystem management when compared to other policy options and measures (Maltby et al., 2013), as also shown in the Mar Menor case.

Assessment of ecosystem services using very simplified methods should be undertaken with caution. For example, some assessments are being carried out by means of remote sensing–based land cover maps, as a proxy of ecosystem types and constant monetary valuations per service of each land cover/ecosystem, leading to a final fixed economic value for a reduced set of general land covers/ecosystems (see, e.g., Cai et al., 2013). This type of approaches has the potential risk of misleading estimations due to the lack of consideration of interactions, dynamic changes, and the importance of context-specific factors. This may hide key processes and services that are not tracked with land cover maps and change along time or present complex interactions with other processes, leading to biased valuations. Moreover, the assessment of land-use changes in terms of ecosystem services by means of a final unique value per cover cannot identify complex trade-offs and the distribution of costs and benefits associated to such trade-offs, which might be relevant for stakeholders. This risk increases in the case of ecosystems less well represented by gross average values, as in the case of wetlands of arid environments. On the contrary, models, particularly context-based or context-adapted models, allow a more detailed consideration of ecosystem processes and their linkages with specific services, leading to reduced risks of bias in the valuation process.

Models are very useful tools to assess the ecosystem services, although it is important to identify the most appropriate approach for each case. For example, Vigerstol and Aukema (2011) review in relation to the assessment of services of freshwater ecosystems the advantages and limitations of well-established hydrological models (SWAT, VIC) and new model tools specifically devoted to assess ecosystem services (InVEST, ARIES), all of them being of interest depending of the specific purpose, available on-site information, and level of expertise, among other criteria.

A context-specific approach may be needed for assessing the services of less common ecosystems as wetlands of arid environments. For example, regarding the drivers of ecosystem change and their services, it is generally recognized that agricultural intensification usually reduces the hydrological flows supporting the wetlands (see, e.g., reviews by Zedler and Kercher, 2005; Maltby and Acreman, 2011; Maltby et al., 2013). However, as shown in the Mar Menor case, in arid environments the agricultural intensification may act in the opposite direction, increasing the hydrological flows reaching the wetlands and altering the oligotrophic and saline conditions of such systems. Therefore, the effects on the wetland ecosystem services of the same driver (agricultural intensification) may be rather different. This points to the need for context-specific approaches. Model outputs cannot be easily transferred between contexts without a reconsideration of model assumptions, structure, parameterization, and intended purpose (Galic et al., 2012). Context-specific or context-adapted models are useful not only for on-site better informed management and decisions regarding the concerned wetland and its services but also as illustrating examples for other cases.

Finally, since the assessment of ecosystem services is particularly useful for management and decision making, there is a need for transparency in the applied modeling tools (Galic et al., 2012) to develop confidence and to contribute to consensus building. The availability of confident modeling tools will help to show the linkages among apparently very different systems, processes, and services, as well as the benefits of more integrated approaches to wetlands, the watersheds they depend upon and—in the case of coastal wetlands—the coastal systems they are linked to. The modeling framework being developed in the Mar Menor tries to contribute to this final aim.

References

Almansa, C., Martínez-Paz, J.M., 2011. Intergenerational equity and dual discounting. Environ. Develop. Eco. 16, 685–707.

Alvarez-Rogel, J., Jiménez-Cárceles, F.J., Egea-Nicolás, C., 2006. Phosphorous and nitrogen content in the water of a coastal wetland in the Mar Menor lagoon (SE Spain): relationships with effluents from urban and agricultural areas. Water Air Soil Pollut. 173, 21–38.

Alvarez-Rogel, J., Jimenez-Carceles, F.J., Roca, M.J., Ortiz, R., 2007. Changes in soils and vegetation in a Mediterranean coastal salt marsh impacted by human activities. Estuar. Coast. Shelf Sci. 73, 510–526.

Álvarez-Rogel, J., Jiménez-Cárceles, F.J., Egea Nicolás, C., María-Cervantes, A., González-Alcaraz, M.N., Párraga Aguado, I., Conesa Alcaraz, H.M., 2009. Papel de los humedales costeros del Mar Menor en la depuración de aguas eutrofizadas: el caso de la Marina del Carmolí. In: Cabezas, F., Senent, M. (Eds.), Estado actual del conocimiento científico. Fundación Cluster-Instituto Euromediterráneo del agua, pp. 321–358.

Astorga, D., Ruiz, J., Prieto, L., 2012. Ecological aspects of early life stages of Cotylorhiza tuberculata (Scyphozoa: Rhizostomae) affecting its pelagic population success. Hydrobiologia 690, 141–155.

Barbier, E.B., 2011. Wetlands as natural assets. Hydrol. Sci. J. 56, 1360–1373.

Bastian, O., 2013. The role of biodiversity in supporting ecosystem services in Natura 2000 sites. Ecol. Indic. 24, 12–22.

Cai, Y.B., Zhang, H., Pan, W.B., Chen, Y.H., Wang, X.R., 2013. Land use pattern, socio-economic development, and assessment of their impacts on ecosystem service value: study on natural wetlands distribution area (NWDA) in Fuzhou city, southeastern China. Environ. Monit. Assess. 185, 5111–5123.

Carreño, M.F., Esteve, M.A., Martinez, J., Palazón, J.A., Pardo, M.T., 2008. Habitat changes in coastal wetlands associated to hydrological changes in the watershed. Estuar. Coast. Shelf Sci. 77, 475–483.

Centro Regional de Estadística de Murcia, 2013. Anuario Estadístico de la Región de Murcia. http://www.carm.es/econet/publica/catalogo_est_sintesis1.html.

Chapelle, A., Duarte, P., Fiandrino, A., Esteve, M.A., Galbiati, L., Marinov, D., Martínez, J., Norro, A., Plus, M., Somma, F., Tsirtsis, G., Zaldívar, J.M., 2005. Comparison between different modelling approaches for coastal lagoonsReport EUR 21817 EN. Institute for Environment and Sustainability. Ispra. Joint Research Centre. European Commission.

Conesa, H., Jiménez-Cárceles, F.J., 2007. The Mar Menor lagoon (SE Spain): a singular natural ecosystem threatened by human activities. Mar. Pollut. Bull. 54, 839–849.

David, M.B., Gentry, L.E., Kovacic, D.A., Smith, K.M., 1997. Nitrogen balance in and ex-port from an agricultural watershed. J. Environ. Qual. 26, 1038–1048.

Dolores, E., Bermúdez, L., Bas, I., Gómez, O., Viuda, E., López, J.I., Pla, M.L., Peñalver, J., 2009. Gestión de recursos pesqueros del Mar Menor. In: Cabezas, F., Senent, M. (Eds.), Mar Menor. Estado actual del conocimiento científico. Fundación Cluster-Instituto Euromediterráneo del agua, pp. 517–540.

Esteve, M.A., Calvo, J.F., 2000. Conservación de la naturaleza y biodiversidad en la Región de Murcia. In: Calvo, J.F., Esteve, M.A., López Bermúdez, F. (Eds.), Biodiversidad. Contribución a su conocimiento y conservación en la Región de Murcia. Instituto del Agua y Medio Ambiente. Servicio de Publicaciones Universidad de Murcia.

Esteve, M.A., Carreño, M.F., Robledano, F., Martínez-Fernández, J., Miñano, J., 2008. Dynamics of coastal wetlands and land use changes in the watershed: implications for the biodiversity. In: Russo, R.E. (Ed.), Wetlands: Ecology, Conservation and Restoration. Nova Science Publishers, New York, NY, pp. 133–175.

Galic, N., Schmolke, A., Forbes, V., Baveco, H., van den Brink, P.J., 2012. The role of ecological models in linking ecological risk assessment to ecosystem services in agroecosystems. Sci. Total Environ. 415, 93–100.

García Pintado, J., Martínez Mena, M., Barberá, G.G., Albaladejo, J., Castillo, V., 2007. Anthropogenic nutrient sources and loads from a Mediterranean catchment into a coastal lagoon: Mar Menor, Spain. Sci. Total Environ. 373, 220–239.

García, J., Rouco, A., García, B., 2001. Evolución del peso económico de la acuicultura marina. Importancia económica de la acuicultura. An. Veter. (Murcia) 17, 41–50.

Gren, I.M., Elofsson, K., Jannke, P., 1997. Cost-effective nutrient reductions to the Baltic Sea. Environ. Resour. Econ. 10, 341–362.
Gustafson, S., Wang, D., 2002. Effects of agricultural runoff on vegetation composition of a priority conservation wetland, Vermont, USA. J. Environ. Manag. 31, 350–357.
Gustafson, A., Fleischer, S., Joelsson, A., 2000. A catchment-oriented and cost-effective policy for water protection. Ecol. Eng. 14, 419–427.
Hernández, A.J., Fernández-Caro, A., 2008. Censo invernal de aves acuáticas de la Región de Murcia, 2003-2008. Memoria. In: Actas IV Congreso de la Naturaleza de la Región de Murcia & I del Sureste Ibérico. Asociación de Naturalistas del Sureste (ANSE), Murcia, pp. 131–158.
Johnson, K., Polasky, S., Nelson, E., Pennington, D., 2012. Uncertainty in ecosystem services valuation and implications for assessing land use tradeoffs: an agricultural case study in the Minnesota River Basin. Ecol. Econ. 79, 71–79.
Jordan, E., Correll, D., Weller, D., 1997. Effects of agriculture on discharges of nutrients from coastal plain watersheds of Chesapeake Bay. J. Environ. Qual. 26, 836–848.
Kampas, A., Edwards, A.C., Ferrier, R.C., 2002. Joint pollution control at a catchment scale: compliance costs and policy implications. J. Environ. Manag. 66, 281–291.
Kronvang, B., Svendsen, L.M., Jensen, J.P., Dørge, J., 1999. Scenario analysis of nutrient management at the river basin scale. Hydrobiologia 410, 207–212.
Lacroix, A., Beaudoin, B., Makowsk, D., 2005. Agricultural water nonpoint pollution control under uncertainty and climate variability. Ecol. Econ. 53, 115–127.
Liu, H.Y., Zhang, S.K., Li, Z.F., Lu, X.G., Yang, Q., 2004. Impacts on wetlands of large-scale land-use changes by agricultural development: the small Sanjiang Plain, China. Ambio 33, 306–310.
Lloret, J., Marín, A., 2009. The role of benthic macrophytes and their associated macroinvertebrate community in coastal lagoon resistance to eutrophication. Mar. Pollut. Bull. 58, 1827–1834.
Lloret, J., Marin, A., Marin-Guirao, L., Velasco, J., 2005. Changes in macrophytes distribution in a hypersaline coastal lagoon associated with the development of intensively irrigated agriculture. Ocean Coast. Manag. 48, 828–842.
Lloret, J., Marin, A., Marin-Guirao, L., 2008. Is coastal lagoon eutrophication likely to be aggravated by global climate change? Estuar. Coast. Shelf Sci. 78, 403–412.
Maes, J., Paracchini, M.L., Zulian, G., Dunbar, M.B., Alkemade, R., 2012. Synergies and trade-offs between ecosystem service supply, biodiversity and habitat conservation status in Europe. Biol. Conserv. 155, 1–12. http://dx.doi.org/10.1016/j.biocon.2012.06.016.
Maltby, E., Acreman, M.C., 2011. Ecosystem services of wetlands: pathfinder for a new paradigm. Hydrol. Sci. J. 56, 1341–1359.
Maltby, E., Acremanb, M., Blackwellc, M.S.A., Everardd, M., Morrise, J., 2013. The challenges and implications of linking wetland science to policy in agricultural landscapes—experience from the UK National Ecosystem Assessment. Ecol. Eng. 56, 121–133.
Martinez Paz, J.M., Martinez Fernández, J., Esteve Selma, M.A., 2007. Evaluación económica del tratamiento de drenajes agrícolas en el Mar Menor (SE España). Revista Española de Estudios Agrosociales y Pesqueros 215 (216), 211–231.
Martínez-Fernández, J., Esteve-Selma, M.A., 2007. Gestión integrada de cuencas costeras: dinámica de los nutrientes en la cuenca del Mar Menor (Sudeste de España). Revista de Dinámica de Sistemas 3, 2–23.
Martínez-Fernández, J., Esteve-Selma, M.A., Robledano-Aymerich, F., Pardo-Sáez, M.T., Carreño-Fructuoso, M.F., 2005. Aquatic birds as bioindicators of trophic changes and ecosystem deterioration in the Mar Menor lagoon (SE Spain). Hydrobiologia 550, 221–235.

Martin-López, B., Gómez-Baggethun, E., García-Llorente, M., 2014. Trade-offs across valuedomains in ecosystem services assessment. Ecol. Indicat. 37 (Part A), 220–228.

Meissner, R., Seeger, J., Rupp, H., 2002. Effects of agricultural land use changes on diffuse pollution of water resources. Irrig. Drain. 51, 119–127.

Mertz, O., Ravnborg, H.M., Lövei, G.I., Nielsen, I., Konijnendijk, C.C., 2007. Ecosystem services and biodiversity in developing countries. Biodivers. Conserv. 16, 2729–2737.

Moreno-González, R., Campillo, J.A., García, V., León, V.M., 2013. Seasonal input of regulated and emerging organic pollutants through surface watercourses to a Mediterranean coastal lagoon. Chemosphere 92, 247–257. http://dx.doi.org/10.1016/j.chemosphere.2012.12.022.

Nemec, K., Raudsepp-Hearne, C., 2013. The use of geographic information systems to map and assess ecosystem services. Biodivers. Conserv. 22, 1–15.

Okruszko, T., Duel, H., Acreman, M., Grygoruk, M., Florke, M., Schneider, C., 2011. Broad-scale ecosystem services of wetlands—overview of the current situation and future perspectives under different climate and water management scenarios. Hydrolog. Sci. J. 56 (8), 1501–1517.

Olhan, E., Gun, S., Ataseven, Y., Arisoy, H., 2010. Effects of agricultural activities in Seyfe Wetland. Sci. Res. Essays 5, 9–14.

Pardo, M.T., Esteve, M.A., Giménez, A., Martínez-Fernández, J., Carreño, M.F., Serrano, J., Miñano, J., 2008. Assessment of the hydrological alterations on wandering beetle assemblages (coleoptera: Carabidae and Tenebrionidae) in coastal wetlands of arid mediterranean systems). J. Arid Environ. 72, 1803–1810.

Parra, G., Jimnenez-Melero, R., Guerrero, F., 2005. Agricultural impacts on Mediterranean wetlands: the effect of pesticides on survival and hatching rates in copepods. Int. J. Limnol. 41, 161–167.

Pérez Ruzafa, A., Aragón, R., 2003. Implicaciones de la gestión y el uso de las aguas subterráneas en el funcionamiento de la red trófica de una laguna costera. In: Fornés, J.M., Llamas, R. (Eds.), Conflictos entre el desarrollo de las aguas subterráneas y la conservación de los humedales: litoral mediterráneo. Ediciones Mundi-Prensa, Madrid, pp. 215–245.

Perez Ruzafa, A., Gilabert, J., Gutiérrez, J.M., Fernández, A.I., Marcos, C., Sabah, S., 2002. Evidence of a planktonic food web response to changes in nutrient input dynamics in the Mar Menor coastal lagoon, Spain. Hydrobiologia 475 (476), 359–369.

Prieto, L., Astorga, D., Navarro, G., Ruiz, J., 2010. Environmental control of phase transition and polyp survival of a massive-outbreaker jellyfish. PLoS One 5 (11). http://dx.doi.org/10.1371/journal.pone.0013793.

Purcell, J.E., Uye, S., Lo, W., 2007. Anthropogenic causes of jellyfish blooms and their direct consequences for humans: a review. Mar. Ecol. Prog. Ser. 350, 153–174.

Robledano, F., 1998. Mar Menor. In: Bernúes, M. (Ed.), Zonas húmedas españolas incluídas en el Convenio de Ramsar. Dirección General de Conservación de la Naturaleza, Ministerio de Agricultura, Pesca y Alimentación, Madrid, pp. 323–334.

Robledano, F., Farinós, P., 2010. Waterbirds as bioindicators in coastal lagoons: background, potential value and recent research in mediterranean areas. In: Friedman, A.G. (Ed.), Lagoons: Biology, Management and Environmental Impact. Nova Science Publishers, Hauppauge, NY, pp. 153–183.

Robledano, F., Esteve, M.A., Farinós, P., Carreño, M.F., Martínez, J., 2010. Terrestrial birds as indicators of agricultural-induced changes and associated loss in conservation value of Mediterranean wetlands. Ecol. Indic. 10, 274–286.

Robledano, F., Esteve, M.A., Martínez-Fernández, J., Farinos, P., 2011. Determinants of wintering waterbird changes in a Mediterranean coastal lagoon affected by eutrophication. Ecol. Indic. 11, 395–406.

Ruiz, J., Prieto, L., Astorga, D., 2012. A model for temperature control of jellyfish (*Cotylorhiza tuberculata*) outbreaks: a causal analysis in a Mediterranean coastal lagoon. Ecol. Model. 233, 59–69.

Saunders, D.A., Wintle, B.A., Mawson, P.R., Dawson, R., 2013. Egg-laying and rainfall synchrony in an endangered bird species: implications for conservation in a changing climate. Biol. Conserv. 161, 1–9.

Schou, J.S., Skop, E., Jensen, J.D., 2000. Integrated agri-environmental modelling: a costs analysis of two nitrogen tax instruments in the Vejle Fjord watershed, Denmark. J. Environ. Manag. 58, 199–212.

Serrano, J.F., Sironi, J.S., 2009. Cuantificación y evolución de la carga contaminante de nutrientes y plaguicidas en aguas del Mar Menor y su relación con los aportes hídricos de la Rambla del Albujón y otros aportes subterráneos. In: Cabezas, F., Senent, M. (Eds.), Mar Menor. Estado actual del conocimiento científico. Fundación Cluster-Instituto Euromediterráneo del agua, pp. 245–284.

Tománková, I., Boland, H., Reid, N., Fox, A.D., 2013. Assessing the extent to which temporal changes in waterbird community composition are driven by either local, regional or global factors. Aquat. Conserv. Mar. Freshwat. Ecosyst. 23, 343–355.

Turner, K., Georgiou, S., Green, I.M., Wulff, F., Barret, S., Soderqviest, T., Bateman, I., Folke, C., Langaas, S., Zylicz, T., Maler, K.G., Markowska, A., 1999. Managing nutrient fluxes and pollution in the Baltic: an interdisciplinary simulation study. Ecol. Econ. 30, 333–352.

Velasco, J., LLoret, J., Millan, A., Marin, A., Barahona, J., Abellán, P., Sánchez-Fernández, D., 2006. Nutrient and particulate inputs into the Mar Menor lagoon (SE Spain) from an intensive agricultural watershed. Water Air Soil Pollut. 176, 37–56.

Verhoeven, J.T.A., Arheimer, B., Yin, C., Hefting, M.M., 2006. Regional and global concerns over wetlands and water quality. Trends Ecol. Evol. 21, 96–103.

Vidal-Abarca, M.R., Esteve Selma, M.A., Suárez Alonso, M.L., 2003. Los Humedales de la Región de Murcia: Humedales y Ramblas de la Región de Murcia. Consejería de Agricultura. Agua y Medio Ambiente, Murcia.

Vigerstol, K., Aukema, J.E., 2011. A comparison of tools for modeling freshwater ecosystem services. J. Environ. Manag. 92, 2403–2409.

Xue, Y., David, M.B., Gentry, L.E., Kovacic, D.A., 1998. Kinetics and modelling of dissolved phosphorous export from a tile-drained agricultural watershed. J. Environ. Qual. 27, 917–922.

Zanou, B., Kontogianni, A., Skourtos, M., 2003. A classification approach of cost effective managent measures for the improvement of watershed quality. Ocean Coast. Manag. 46, 957–983.

Zedler, J.B., Kercher, S., 2005. Wetland resources: status, trends, ecosystem services, and restorability. Annu. Rev. Environ. Resour. 30, 39–74.