MedWet

RESTORATION OF MEDITERRANEAN WETLANDS

G. C. ZALIDIS T. L. CRISMAN P. A. GERAKIS

E D I T O R S

HELLENIC MINISTRY OF THE ENVIRONMENT, PHYSICAL PLANNING AND PUBLIC WORKS

THE GOULANDRIS NATURAL HISTORY MUSEUM
GREEK BIOTOPE/WETLAND CENTRE

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This volume was published with the financial support of the Hellenic Ministry of the Environment, Physical Planning, and Public Works, in the framework of the participation of Greece in the Mediterranean Wetlands Committee and in the MedWet team. It is one of the several contributions of Greece to the collective efforts of the Mediterranean countries to conserve and sustainably manage the natural heritage which is also cultural heritage, and to exchange experiences and promote collaboration.

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MedWet

The Mediterranean Wetlands Initiative (commonly known as MedWet), started in 1991 with the participation of five Mediterranean governments, the Ramsar Convention and a few institutes and organizations. Participation gradually increased. Presently the participants are all governments of the Mediterranean countries, the Palestinian Authority, and the governments of some other countries which, although not having direct access to the Mediterranean coast, their biodiversity is connected with Mediterranean wetlands. Several non-governmental organisations, international conventions, and scientific institutes also participate.

Medwet works to implement the Mediterranean Wetlands Strategy which was agreed upon in Venice in 1996 by the Medwet partners. The mission of this Strategy is "*to stop and reverse the loss and degradation of Mediterranean wetlands as a contribution to the conservation of biodiversity and sustainable development in the region*".

Medwet activities include the development of approaches and methods on: inventories and mapping, monitoring, training, public awareness, diffusion of research results, sustainable management etc. It enjoys the moral and material support of the European Commission.

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PREFACE

"*The history of the Mediterranean trace-passes through wetlands*". These are the very first, and highly significant, words of this very useful piece of work on the restoration of wetlands. Because these few words encapsulate the relevance of wetland ecosystems in the region, and the need to bring them back, as much as possible, to what they have been in the past.

The Convention on Wetlands (Ramsar, Iran, 1971) has been paying significant attention to wetland restoration, with several resolutions and recommendations of the Conference of the Contracting Parties (COP) urging member countries to undertake actions in this direction. Our resolutions on inventory have insisted upon the need to register those weltands in need of restoration. And the most recent COP, held in May 1999, adopted Resolution VII.17 on *Restoration as an element of national planning for wetland conservation and wise use*, in which the COP, *inter alia, "calls upon all Contracting Parties to recognise that although restoration or creation of wetlands cannot replace the loss of natural wetlands, and that avoiding such loss must be a first priority, a national programme of wetland restoration, pursued in parallel with wetland protection, can provide significant additional benefits for both people and wildlife, when the restoration is ecologically, economically and socially sustainable".*

The Resolution also *"urges Contracting Parties to produce information about wetland losses, including an assessment of the lost processes, functions, composition and values of wetland areas. This information should include data about the restoration potential of these sites and the full benefits of restoration, including identification, at all appropriate levels and using standardised protocols for data gathering and handling of sites that are a priority for restoring for the benefit of people and the natural environment".*

The work undertaken by EKBY and assembled in this publication constitutes a very useful tool for the Mediterranean countries (all of which are members of the Convention), and other interested parties, for implementing the Ramsar COP Resolution cited above.

I very much hope that this tool will encourage those responsible for wetland management in the Mediterranean to identify additional restoration projects in the region. A number of important projects have been launched in the European Union countries of the basin, but there is a need to undertake similar initiatives all over the Mediterranean, wherever wetlands have been lost or degraded at a significant scale, mainly at times where there was not a clear understanding of the functions and values of wetlands. Today, we have made important progress in this direction, but the pressure on the remaining wetlands is still very strong, and the restoration of their functions is not yet sufficiently seen as an important activity that deserves important investment.

At the next meeting of the Ramsar COP, in November 2002, it is foreseen that Parties will consider and adopt a resolution on *Principles and guidelines for wetland restoration*. In the Guidelines, Parties would be recognising that *"although there is an increasing interest in wetland restoration and opportunities are widespread, the efforts to restore wetlands are still sporadic, and there is a lack of general planning at the national level. Individuals and organizations interested in restoration often work in isolation and without the benefit of experience gained on other projects"*.

These new Ramsar Guidelines will be offering a series of general principles on wetland restoration and a road map on how to go from establishing project goals, objectives, and performance, to implementing a monitoring programme, reconsidering the original objectives and taking remedial actions.

EBKY is in many ways anticipating this initiative of the Convention by already offering countries an extremely useful tool of action. For this, we are once more very grateful to the Centre for its contribution to the progress of wetland knowledge and the effective management in the Mediterranean.

Delmar Blasco
Secretary General
Convention on Wetlands (Ramsar, Iran, 1971)
Gland, Switzerland, 23 April 2002

GENERAL INTRODUCTION

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The history of Mediterranean trace-passes through wetlands. From the Nile river of the Pharaohs to the Axios river of Alexander the Great and from the Dead Sea in the Jordan Valley to the Darro river in the Alhambra of Granada, wetlands have been places where the Mediterranean civilizations thrived. During the last century, the increasing population and the dominance of unsustainable policies pertaining to human activities such as crop and animal farming, tourism, urban and industrial development, fishing, energy use, etc. have caused great losses to Mediterranean wetlands. About two thirds of wetland areas were drained, while most of the remaining areas continue to deteriorate. On the other hand, a positive attitude of Mediterranean societies towards wetland conservation started to emerge some decades ago as science revealed the great importance of wetland functions and values for man and nature.

Functions and values of Mediterranean wetlands

Wetland ecosystems, throughout the history of Mediterranean civilizations, were strongly related to human welfare and the conservation of natural resources, notably water, soil, and genetic resources. Traditional economic activities were based around wetlands. Wetland values include all commodities and services provided by wetlands to mankind. These values may either be directly exploitable, such as drinking and irrigation water, feed for livestock, and places suitable for tourism, or indirectly beneficial, such as biodiversity, opportunities for recreation, scientific study, and education.

A wetland may have several of the following 20 values: biodiversity, drinking water, irrigation water, hydropower, fishing, feed for farm animals, game, wood and reed production, salt production, sand production, science, education, culture, recreation, protection against floods, protection against erosion, improvement of water quality, improvement of topoclimate, transportation, treatment of human ailments. Few countries of the Mediterranean have inventoried their wetlands as to the number of values each wetland possesses, the level of each value and degree of use of each value. It is noteworthy that the levels of values and the degree of use may substantially change even from one year to another.

Whether direct or indirect, wetland values derive from wetland functions. The latter can be defined as sets of physical, chemical, and biological processes that take place within wetland ecosystems. These processes are the combined result of numerous interactions between the structural components of wetlands (soil, water, vegetation, etc.) and between wetlands and the surrounding environment, i.e. the characteristics of the watershed area.

By reviewing the literature one can find several definitions of wetland functions and values. Some authors list some functions as values and vice-versa. One has to keep in mind, however, that the concept of wetland functions has been developed by scientists in order to facilitate the understanding and management of wetland ecosystems. Clearly, the following seven terms are widely recognized as denoting the main wetland functions:

1. Water storage.
2. Food web support.
3. Groundwater recharge.
4. Sediment and toxicant trapping.
5. Nutrient removal and transformation.
6. Floodwater attenuation.
7. Shoreline stabilization.

A question is sometimes raised on whether the first term, i.e. water storage, should be called a function that the wetland performs or if it is a pre-supposition for the very formation and existence of a wetland (G. E. Hollis 1990, personal communication). While the question is theoretically interesting and all arguments may be valid, the recognition of water storage as a distinct function is advantageous from the point of view of wetland management.

The number of functions, their relative importance and the degree of their performance differ from wetland to wetland. Each wetland is a unique ecosystem with characteristics dependent on watershed, and location in the landscape, climate, and numerous other factors. As a result, wetlands differ in the number of functions they perform and in the degree of performance of each function.

In most cases, wetland functions are strongly interrelated and therefore degradation or restoration of a function may have negative or positive effects, direct or indirect, on other functions.

The degradation of Mediterranean wetlands

The overall percentage of Mediterranean countries covered by wetlands masks large differences, particularly with respect to the Northern European countries. For example, the wetland areas in Spain cover only 0.3% of the country's total area, while in Sweden, a cooler and wetter country of Northern Europe, they cover 28%. In Mediterranean countries, therefore, the important hydrological functions provided by wetlands are being delivered from a very small proportion of the territory, making these resources particularly important and at the same time vulnerable to careless planning and management.

Ancient myths in the Eastern Mediterranean and the Middle Eastern countries make reference to wetland drainage plans. However, going back in history, perhaps the first large-scale wetland drainage project was constructed in the Roman period. For centuries, the Pontine Marshes were responsible for widespread malaria epidemics in central Italy. The Pontine Marshes covered about 70,820 ha south of Rome, between Cisterna and Terracina. The Roman emperors Trajan and Theodoric, and later Pope Sixtus V, drained the parts above sea level by digging drainage ditches. In 1513, Leonardo Da Vinci traveled to Rome where he entered the service of Guiliano de Medici, the brother of Pope Leo X. There, between 1513 and 1515, he drew plans for draining the Pontine Marshes which were the basis of a large-scale reclamation project financed by the Medici family. In the 1930's, the rest of the marshes were drained by a system of dikes and pumps and the use of Eucalyptus trees.

Although Mediterranean wetlands are threatened by numerous human activities, until today there has been no detailed study of wetland degradation and loss in the whole of the Mediterranean. However, it is estimated that over the last 100 years 28% of the Tunisian wetlands have disappeared. The watershed of the Medjerda has the highest loss of wetland area, i.e. 84%, and the main causes of loss has been drainage (14,353 ha), urbanization (3,341 ha), agricultural encroachment (1,353 ha) and upstream dams (500 ha). Spain has lost 40,000 ha of wetlands in recent years, three-quarters of which were connected to aquifers. In Roman times, 10% of Italy (3 million ha) were wetlands, by 1991 this had diminished to only 300,000 ha (Anonymous 1996). At a national level Greece drained 60% of its wetland area during 1920-1962 (Gerakis 1992) and the percentage was higher in its prefectures in East, Central, and Western Macedonia, where the total lake area was reduced from 58,600 ha to

36,400 ha and total marshland area from 98,600 ha to 5,600 ha (Psilovikos 1992). Even some rivers were drained (e.g. Gallikos river in Greece) by diverting their waters for urban and agricultural use.

A French Government study has found that 86% of the most important wetland sites in France have been degraded as a result of official public policies imposed during 1964-1994 which encouraged drainage and conversion of wetlands. As a result of such actions, wetlands are now probably the most threatened ecosystems in Europe (Baldock et al. 1984).

The overall problem of Mediterranean wetlands stems from the policies developed to satisfy the increasing fresh water needs for drinking, irrigation, industry, and hydropower.

Drinking water is in short supply in most of the coastal zone of the Mediterranean due to the climate and the fact that 130 million live in that zone or close to it and millions of tourists visit that zone mainly in the summer. The situation will become more serious in the future in view of the population increases expected, especially in the countries of the African coast.

Irrigated agriculture is the major consumer of water in the Mediterranean (70-80% of the total water consumed). Most of this water comes from surface sources. Unsustainable use of irrigation water is the primary cause of deterioration of several wetlands in Greece (Gerakis and Kalburtji 1998) and other countries. For example, in Lake Oubeira in Algeria all water was pumped out of the lake during the 1990 drought (Pearce 1996). It is perhaps encouraging that some irrigation specialists (e.g. Papazafiriou 2000) claim that up to 40% of irrigation water can be saved by improved management of irrigation schemes based on modern technology. Wisely managed irrigation schemes may become a major asset to biodiversity.

Hydropower dams are found in many rivers flowing in the Mediterranean. Most of them are multipurpose dams, that is, they also store water for irrigation and act as flood control devices. Their disruptive effects on downstream wetlands are well known through the most prominent example, i.e., the Aswan dam on the Nile. Other less widely known examples refer to the Rhône river in France, and the Nestos and Acheloos rivers in Greece. Current plans for new small dam construction in Mediterranean rivers is a controversial issue due to conflicts of interest and to lack of reliable information of the true long-term costs and benefits of a dam. In older damming efforts, neither the loss of ecological (e.g. biodiversity) and economic (e.g. fishing, recreation) values of downstream areas nor the acceleration of coastal erosion due to the prevention of sediment deposition were viewed as cost parameters.

Industry is a minor user of water in most Mediterranean countries. It is generally more a polluter of water bodies than a user of water. There are cases, however, that industrial plants use large amounts of very high quality water for cooling purposes only, thus putting pressure on the, usually, limited drinking water supplies of nearby cities.

The direct and indirect flow of pollutants into wetlands is a ubiquitous phenomenon all over the world and well documented in numerous Mediterranean wetlands. The detrimental effects become more serious in the summer when the volume of wetland water decreases due to high evaporation (for all wetlands) and to increased abstraction of water for irrigation (for freshwater wetlands only). Although most countries in the Mediterranean have adopted strict anti-pollution regulations, enforcement in most cases still leaves much to be desired. A controversial issue is the relative role of non-point pollution from the agroecosystems of the wetlands' perimetric zone. The controversy is due to the difficulties in obtaining direct evidence from local field observations. On the other hand, the indirect evidence from United States studies and from simulation models should motivate the adoption of sustainable practices in those agroecosystems. Measures to encourage such practices have been in effect by the European Union since 1992.

Arresting and reversing the loss and degradation: sustainable use and restoration

The negative forces exerting their influence on the Mediterranean wetland wealth are still strong. This influence, however, is seldom expressed today as complete drainage but as degradation of the functions of the existing wetlands.

Positive forces have started to rise in most countries of the Mediterranean during the last three decades. One piece of evidence is the several international environmental protection conventions which have been signed by most countries of the Mediterranean, such as those on the transboundary movements of hazardous wastes, conservation of wildlife, desertification, climatic change, biodiversity, and, most notably, the convention on wetlands (known also as the Ramsar Convention) which is the only international agreement referring to one particular ecosystem type.

Other evidence is the multiplication of public awareness campaigns and studies and the establishment of three more scientific institutes (EKBY in Greece, SEHUMED in Spain, and ICN in Portugal) for Mediterranean wetlands in addition to the much older and better known Station Biologique de la Tour du Valat in France.

Sustainable use of the existing wetlands is the road to their preservation. Sustainable use may often suffice to bring back some lost or degraded functions. There are many cases, however, where additional interventions, commonly grouped under the term restoration, are necessary.

Ecology, unlike other much older scientific disciplines (e.g. physiology, systematics, genetics), is still characterized by a high diversity of opinions on the definition of many terms; ecosystem restoration is one of those terms. One should view this diversity of opinions on restoration and related terms not as an obstacle to useful work for wetlands but as an opportunity to advance ecological theory. Nevertheless, communication among scientists is enhanced if controversial terms are explained by the authors or editors of pertinent studies and reports.

In the context of this publication, wetland restoration is defined as the creation, re-establishment, or enhancement of wetland functions to a self-sustainable level. Thus, the term restoration refers not only to lost or degraded wetlands but also to the wetlands that are created in dryland areas where wetlands never previously existed. The philosophy of this definition originates from the enormous need of the Mediterranean countries to recover, using any means, as much as possible of the wetland wealth that they have lost. One approach is to upgrade the existing wetlands, another is to revive the drained wetlands and another is to construct new wetlands in new places when even the partial revival of the lost wetlands in their original locations is impossible.

Some additional explanations are useful. The authors of this publication often find themselves in the need to use other terms too when clarity requires it. For example, wetland re-creation is used to denote the construction of a wetland on the site occupied in the past by a drained one. This does not mean that it is ever humanly possible to exactly revive the drained wetland with the same area, structure, and functions. Thus re-creation means partial creation of a wetland in a place where a lost one existed.

Wetland restoration can be seen from different angles: as a tool in conservation and management, as a means to increase wetland hectarage, as an approach to promote sustainable regional economic development, as a prerequisite to meet the challenges set by the Water Framework Directive for most water districts of the European Union, and as a branch of the disciplines of Ecology and Hydrology. Moreover, restoration projects can be seen as one of the strongest and more easily comprehensible arguments against wetland loss and degradation. Most of the wetland restoration projects in the Mediterranean were demanded by the children of the people who had drained wetlands.

A synopsis of various restoration projects in Mediterranean countries was made by Zalidis et al. (1999) and a description of the perspectives of Mediterranean wetlands in the turn of the new century may be found in the work of Papayannis and Salathé (1999).

The origin of this publication

Wetland projects specifically aimed at restoration is a fairly recent development in the Mediterranean. Most of these projects were and are carried out in the European Union countries because these countries have higher availability of funds. However, all the countries of the Mediterranean have expressed through their participation in the Mediterranean Wetlands Initiative (MedWet) their decision to promote the undertaking of many more restoration projects all over the Mediterranean.

The first demonstration of the MedWet partners decision to promote restoration was in the form of a technical session on this subject. The session was held during the 1st Meeting of the Mediterranean Wetlands Committee which was hosted by the Greek government in Thessaloniki, Region of Central Macedonia, Greece, in March 1998. A short technical bulletin was produced 1 year later (Kontos et al. 1999) with the papers presented at that session. The Committee identified the need for more collaborative work on restoration recognizing the common characteristics and problems of Mediterranean wetlands and the advantages from the exchange of experiences from past and current restoration projects in various Mediterranean countries.

The Hellenic Ministry of the Environment, Physical Planning and Public Works, in the framework of its participation in the Mediterranean Wetlands Committee, decided to contribute to the dissemination of wetland restoration know-how in the Mediterranean by funding this publication whose preparation is a collective endeavor.

The publication aims to provide up-to-date information on theoretical and applied aspects of wetland restoration in the Mediterranean. It offers broad Mediterranean representation, but also includes chapters written by restoration experts from other parts of the world, in order to provide wider experience. The publication focuses on scientific and technical issues, taking into account that restoration is an open venture that involves local community stakeholders, attempts to analyze the restoration efforts made in the Mediterranean socioeconomic and ecological environment, and outlines the lessons drawn from these efforts.

Restoration includes both prevention and therapy. For this reason, it should be incorporated into local, national and international policies for wetland conservation, and from that perspective this publication is addressed to scientists, wetland managers, public administrators, non-governmental organizations, and decision-makers.

The editors are convinced that this volume contains particularly useful material for government officials involved in wetland management and for university post-graduate students of Ecology, Hydrology, Environmental Science, and Engineering by complementing the more theoretical and extensive books on restoration previously published.

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PART 1

PHYSICAL, CHEMICAL, AND BIOLOGICAL ASPECTS OF RESTORATION

ECO-HYDROLOGY AND WETLAND RESTORATION

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Introduction

The ability to restore a degraded wetland, depends on a number of factors including the type of wetland, the ecological functions of interest, the type and degree of degradation, the land uses in the entire watershed, and the ability to establish and maintain appropriate hydrologic conditions (Kusler and Kentula 1990, Kentula 2000). Although many restored wetlands are successfully providing desired results, there have been some cases of failure generally caused by a lack of proper hydrology (Mitch and Gosselink 1993).

Mitsch and Gosselink (1993) characterize wetland hydrology as the most important factor in the ecology, management and restoration of wetlands, and probably as the single most important determinant of the establishment and maintenance of specific types of wetlands.

Wetland hydrology takes into account not only the water budget of the individual ecosystem, but also the water balance within the watershed where it resides. Hydrologic inflows into the wetland include precipitation, flooding rivers, surface flows, groundwater and tides (in the case of coastal wetlands), while hydrologic outflows include evapotranspiration, as well as surface and groundwater outflows. All these hydrologic components influence vegetation, fauna, most wetland functions, the shape, size, depth and even the location of a wetland; they are the driving forces transporting energy and nutrients to and from the wetland (Kusler 1988).

The interest in wetland restoration in Mediterranean countries stems from the fact that those countries have lost not only a large proportion of their wetland area, but also a significant part of invaluable economic resources.

Wetland hydrology is the key factor controlling the structure and functions of any wetland and therefore is a vital consideration for successful wetland restoration (Zalidis et al. 1999a). It also defines the hydroperiods of the restored wetland, characterized as "the hydrologic signature", that will directly affect wetland vegetation and the performance of ecosystem functions.

This chapter outlines the role of hydrology in the performance of wetland functions, points out the importance of setting clear objectives in restoring wetland hydrology and using appropriate success criteria. Key hydrological aspects of wetland restoration such as hydrological interactions between the wetland and its watershed, setting of reference levels, etc. are also discussed.

Particularities of Mediterranean wetlands

A large number of Mediterranean wetlands have been managed as water reservoirs and wastewater disposal areas, altering their natural functions (e.g. pollution, excessive pumping). On the other hand, during the first half of the 20th century, about one third of the Mediterranean wetland area had been drained, in order to control malaria and expand land for agricultural, industrial and urban purposes (Zalidis et al. 1999b).

Among the most important factors characterizing the climate of a region and affecting the hydrological cycle in a watershed, are the temperature and the rainfall and their distribution throughout the year. According to Acreman (2000), in the Mediterranean basin there is a difference (in terms of rainfall) between the west and east coasts, and there is an inter-annual variability in rainfall and sporadic occurrence of extreme events. The Mediterranean climate (which is characterized by hot dry summers and mild wet winters), along with the fact that water demand rises sharply in the summer, may adversely affect natural wetlands.

Portugal, Spain, France, Italy, and Greece, which are member states of the European Union, have adopted the Water Framework Directive (WFD) 2000/60/EC for community action in the field of water policy (Official Journal of the European Communities 2000). The WFD is considered an instrument that will have far-reaching consequences for the future management of water and aquatic ecosystems throughout Europe. European countries are obliged to establish integrated river basin management plans, which depend critically on reconciling all natural processes and human activities that influence the water cycle in a given river basin (World Wide Fund for Nature 2001). The implementation of this WFD is expected to have specific benefits, including improvement of ecological quality of European freshwater and coastal water ecosystems, and reduction of water pollution.

Assessment of wetland functions to identify causes of degradation

The objective of a restoration project is not only to restore or even enhance degraded wetland functions, but also to eliminate or reduce the causes of degradation directly or indirectly affecting the wetland (water pumping, untreated wastewater discharge, intensive agriculture etc.). For this reason, and under the light of the need to implement a successful integrated restoration plan that will drive the establishment of a self-sustainable wetland, two parameters should always be considered in the initial stage of the planning phase: 1) to identify which functions and to what extent have to be restored, in both the wetland and its watershed, and 2) to identify and eliminate the causes of degradation. To cope with both, as it has been analyzed in other parts of this publication, assessment of wetland functions in response to hydrology is an essential tool.

Setting the potential restoration objectives

Project goals and objectives should reflect information gathered during exchanges with stakeholders (such as environmental and socioeconomic characteristics of the area in question), while benefits and possible negative impacts should be effectively communicated to the stakeholders and other partners (White and Bayley 1999).

Success of a restoration project depends on clear and realistic objectives (taking into account site constraints and financial limitations). The watershed in which a wetland resides both positively and adversely influences the hydrology of the wetland. Positive influences may include water supply from surface flows and groundwater, while adverse influences may include interruptions and fluctuations in the water supply due to upstream diversions, or changes in groundwater level, and water quality changes (Vance et al. 1987).

A water supply must be available to meet the needs of the wetland. The hydrology of the restored wetland is defined as the rate, path, volume and timing of inflow and outflow, including duration, frequency, timing and depth of flooding, ponding or saturation. A frequent goal is that the overall hydrologic variability of the restored wetland must approximate the conditions that existed before alteration, such as dynamic and static water levels and soil saturation conditions.

It should also be stressed that restoration does not imply the permanent reinstatement of a wetland to exactly the same conditions that prevailed before degradation. The primary aim of restoration projects should be to create self-sustainable ecosystems resilient enough to current conditions and characteristics of the whole watershed. The restorative measures imply the re-establishment of wetland functions in response to the contemporary ecological, and socioeconomic conditions of the specific area (Zalidis et al. 1999b).

Significant landscape modification takes significant input of time and funds. For that reason, before any restoration is pursued the purposes should be clearly identified. Any wetland function that has been found degraded during the analysis phase has to be restored to the desired reference level. Therefore, restoration objectives often include ecosystem functions, and selected values such as aesthetics, protection against floods, and water quality improvement.

Ecosystem functions

The link between hydrology and ecosystem function is tight, but poorly understood. Furthermore, it is uncertain whether optimizing for ecological functions simultaneously optimizes for protection against floods and/or water quality purposes. This question should be a major research focus in the near future.

Aesthetics

In regions where wetlands are part of the undisturbed landscape, there may exist significant social or economic pressures to reestablish the natural order by restoring wetlands in order to reestablish the character of the landscape. Such efforts, while well meaning, may not always involve sufficient consideration of the wetland functions. In particular, the anthropogenic aesthetic bias tends to abhor the natural variation in water levels that are common to most wetlands. Often, efforts with aesthetics as the major objective involve the idea of a design pool level. Ultimately, such wetlands are likely to require significant maintenance in order to remove the sediment accretion that oxidation would otherwise destroy. Furthermore, while careful planting can create a natural look under constant pool conditions, ecological functions are certainly compromised. From a hydrologic standpoint, constant pool conditions typically require an overall excess of water to the wetland along with an outlet control structure.

Protection against floods

Riparian and upland wetlands help to attenuate downstream flooding by providing storage upstream. Restoration of flood protection properties does not, in the short term, require the performance of all the ecological functions of the wetland. However, long-term soil stability and other issues suggest that an ecologically functioning wetland will serve the long-term flood protection value better than a wetland that may fall to an unacceptable state. Hydrologic modeling may be used to design restoration efforts for flood protection.

Water quality improvement

Even small changes in wetland hydrology can significantly affect the physical and chemical properties of a wetland, such as nutrient availability, degree of substrate anoxia, soil salinity, sediment properties and pH. Because hydrology plays a vital role in the structure of a wetland ecosystem, particularly by acting as the main pathway in which nutrients are transported into and out of the system, the vegetation and species composition are significantly affected when natural or man made hydrologic alterations occur, thus changing species composition and declining ecosystem productivity (Mitsch and Gosselink 1993).

Similar to the arguments made for flood protection, riparian and upland wetlands can serve as important “filters” for rivers and lakes. Land alterations that prevent the proper performance of functions of such wetlands typically involve short-circuiting or bypassing that prevent the uptake and/or alteration of significant contaminants including, but not limited to biochemical and chemical oxygen demand agents, nutrients, and heavy metals. Restoration of the functions which are responsible for the water quality of the wetland requires, at the least, the restoration of hydrology.

Data collection for simulating water and wetland nutrient dynamics

During all stages (planning, implementation, and monitoring), a restoration project should be considered as an experiment. No two wetlands are identical, which in turn means that not even two restoration projects have the same characteristics and particularities. Environmental status, land uses, and social and economic situation of the surrounding region, need to be considered.

However, during a restoration project it is crucial to collect all available data and use the most appropriate techniques (historical analysis etc.), in order to simulate water and nutrient dynamics of the wetland before degradation, and set the new desirable status of the restored wetland, taking into account any environmental and socioeconomic change that has taken place.

Existing hydrologic conditions and the interactions between the wetland and its watershed

Only by considering all of the processes for a specific watershed both spatially and temporally, can the functions of wetlands within the watershed, and their possible degradation, be understood (Ripl et al. 1994). An assessment of the existing hydrologic conditions should be investigated, in order to quantify the spatial and temporal distribution of the water, both in the wetland under restoration and throughout its watershed.

Comparison of the existing and expected water balances for a watershed is a useful method for identifying water supply problems, assessing impacts of proposed engineering interventions and estimating the magnitude of unknown hydrologic components, such as groundwater flow and infiltration losses (Hayes et al. 2000).

In cases where limited data exist, a monitoring program must be established, along with the use of physically based hydrological models which may help in estimating the pre-degradation hydrologic characteristics of the wetland (as referred by Al-Khudhairy in this publication).

Soil and sediment properties

Nutrients (especially nitrogen and phosphorus), along with sunlight, carbon dioxide and water, are the essential factors for plant and animal growth in every aquatic ecosystem. Nutrient sources for a specific wetland include point and non-point surface water sources (such as urban and/or agricultural wastewaters), groundwater inflow and nutrients released from the accumulated organic sediments; they are generally used along with other parameters to explain eutrophication in aquatic ecosystems.

Nutrients are accumulated in the sediments and bound there. However, overloading by nutrients from external sources results in an increased production of plant matter in a wetland, which in turn may cause a spectrum of processes at the water-sediment interface. Thus, accumulated organic sediments in a wetland, may act as sources or sinks for nutrients and organic pollutants, since degradation processes generally enhance the development of anoxic events and the release of nutrients from sediments (Bjork 1994, De Casabianca et al. 1997).

Under a specific restoration project, there is a strong possibility of areal expansion of the restored wetland, which means expansion of flooded soils that could release nutrients (N, P, S etc.) to the water. Thus, sediment and soil chemical analysis should provide necessary data that must be combined with the expected hydroperiods of the restored wetland, in order to provide the expected new status of the nutrient dynamics of the wetland.

Setting the reference level

After the collection and interpretation of all available data, the designation of the reference level follows. The reference level refers to the functions that the wetland performed before degradation, or to the target-functions that the restoration designer puts as a target, under the prevailing environmental and socioeconomic conditions of the whole watershed. The driving force to set the reference level is the need to re-establish a new self sustainable wetland ecosystem that will perform its desirable functions and serve nature and humanity, with no need of external assistance.

Defining reference conditions for a wetland under restoration is probably one of the most difficult tasks. Useful techniques for this effort, include historical analysis, modeling, ecological analysis and multivariate cluster analysis, as proposed by Kondolf and Larson (1995), Camp et al. (1997), Sagers and Lyon (1997) and Harris (1999).

Taking into account that historical data are usually not available, appropriate natural analogs should be identified.

A logical diagram of steps needed for setting the reference level is presented in Figure 1.

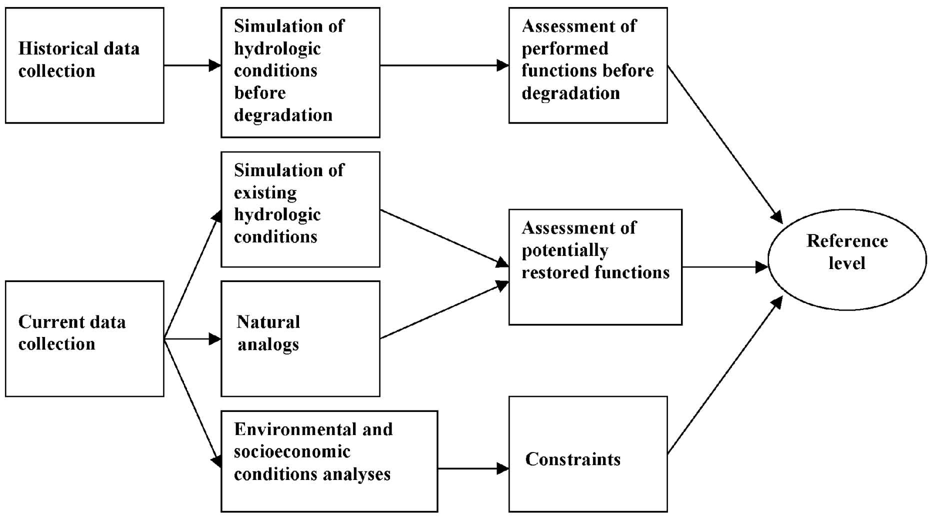

Figure 1. Setting the reference level.

Principles of water allocation

The interactions between the wetland ecosystem and the other water resources in the watershed with which it is hydraulically or ecologically connected must be identified. Restoration of wetland ecosystems requires an integrated approach at the watershed level in order to eliminate or at least minimize the sources of degradation and to establish desirable wetland functions. Establishment of constructed wetlands for water purification purposes, land use regulation and wise management of available water resources are valuable tools for the restoration of degraded wetlands. The management of the existing water resources in the watershed should be governed by the following principles as they were suggested by the Scientific and Technical Review Panel of the Ramsar Convention:

- *Sustainability as a goal.* Adequate water has to be provided to wetlands to sustain the functioning of these ecosystems for the benefit of future generations.
- *Clarity of procedure.* The procedure by which decisions are made on the allocation of water must be clear to all stakeholders.
- *Equity in participation and decision-making factors.* There must be equity for different stakeholders in their participation in water allocation decisions.

- *Credibility of science*. Scientific methods used to support water allocation decisions should be credible and supported by review from the scientific community.
- *Transparency in implementation*. Once procedures for water allocation decisions are defined and agreed, it is important that they are implemented correctly.
- *Flexibility of management*. It is essential that an adaptive management strategy be adopted.
- *Accountability for decisions*. Decision-makers must be accountable.

The extensive use of wetland water for agricultural, municipal and industrial purposes has become a threat for numerous wetlands suffering from water deficiency. Changing water chemistry of several Mediterranean rivers is one of the most important environmental pressures, and often results from overexploitation through direct pumping, irrigation networks and dam construction.

A decision support system could be a useful tool in order to: a) find the spatial distribution of crop pattern (which meets the water demand constraints), b) optimize land uses, and c) provide an interactive visual and calculating tool. It may also help the decision maker to understand the spatial components of the environmental problem, to create alternative scenarios of land use changes and select the most appropriate restoration scenario.

Hydrology- driven success criteria

Evaluation of restoration projects has been conducted in a number of ways, while many variables have been used for that purpose. From a hydrologic point of view, the most frequently used variables are the water budget and timing.

A first cut at defining a hydrologic success criterion might include a water budget comparison of the target wetland with appropriate natural analogs in close proximity. How do the seasonal hydrographs compare? Among the first problems to arise is the paucity of data for most wetland systems with which to compare restored areas. Compounding this is the climatic variability that operates at an entirely different timeline than typical decision makers employ.

For water budgeting to work, the major parameters must be well measured and/or modeled at a site. These include precipitation, evapotranspiration, surface water exchanges, groundwater exchanges and, of course, changes in storage. The first two require meteorological data, the third flow data, and the last bathymetric data. Typically, the fourth, groundwater exchanges, are treated as the residual process that is deduced by bringing the remaining data into concordance. More advanced work has recently focused upon groundwater/wetland interactions and soil-zone storage of water. Nevertheless, true measurement of groundwater exchange is not really possible without great effort, such as application of a tracer test.

As water drives all wetland processes, it should be monitored in all significant wetlands. In systems of isolated wetlands in close proximity, key wetlands may, of course, be chosen as representative of a larger number. At a minimum, water levels in the wetlands of interest (including controls) should be monitored. For remote systems, this is conveniently performed by installation of a stilling well instrumented with a pressure transducer connected to a dedicated data logger. Such setups may be downloaded as often as necessary. Current technology allows for tens of thousands of data points to be collected within data loggers, so the download frequency can be quite low, if desired. However, downloading visits are also opportunities to verify the functioning and recalibrate pressure transducers, and should therefore not be too infrequent. Wireless technology has enabled sites to be monitored on a real-time basis, where resources are sufficient.

Depending upon the hydrologic regime, it can also be desirable to monitor the groundwater below and beyond the wetlands of interest. Wells installed at the appropriate depths (corresponding to the local hydrogeological model) and instrumented as above are appropriate for such purposes. Wells installed through the wetland should be installed with great care in order to prevent short-circuiting between the groundwater and the wetland water. Grouting such wells with a swelling clay, such as bentonite, at the organic sediment horizon has proven successful for this purpose. Direct comparison of the water level of the wetland to the level of groundwater will reveal if the wetland operates under discharge or recharge conditions, or if it oscillates between the two. For tidally influenced systems, the appropriate measures should be taken to capture the tidal signal.

Pressure transducers installed in stilling and groundwater wells yield water levels that can be related to a local datum, such as the ground surface. In order to tie in information among sites or other hydrologic features, additional effort must be expended. For these purposes, traditional surveying or global positioning system (GPS) equipment may be employed. For large-scale studies, the development of a geographical information system (GIS) database can greatly aid the management and interpretation of water level data.

To supplement the water level data, wetland eco-bathymetries can record both the land surface elevation and the dominant ecological features (as type-obligate, facultative etc. or species). When brought into GIS, such information can be married with the hydrologic data to produce a composite model of the wetland system. Weather data can be collected on site or gathered from remotely sensed data, such as radar estimates of rainfall and satellite-based estimates of evaporation.

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THE ROLE OF THE SOIL IN RESTORING WETLAND FUNCTIONS

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Introduction

The three major components of wetlands are water, vegetation, and soils. Because of temporal and spatial variability in hydrology and vegetation, most wetlands are considered to have periodically saturated or flooded soils as a common feature (Reddy and Patrick 1993). Wetland soils have variable organic content, structure, texture and profile development. They are commonly differentiated into organic (OM > 20%) or mineral (OM < 20%) types. As a result of differences in organic content, physico-chemical properties of organic soils differ from those of mineral soils by having lower bulk density and hydraulic conductivity, but higher porosity, water holding and cation exchange capacity (Mitsch and Gosselink 1993, Jacobsen 1990).

Whether mineral or organic, a common feature of all wetland soils is that they are at least periodically saturated with water. In Soil Taxonomy, there are three water saturation-related types: poor drainage, aquic moisture regime and hydromorphic (gley) soils. Although they closely correspond, they do not yield the same demarcation of wetlands nor do they always identify soil wetness or anaerobic conditions (Brinkman and Van Diepen 1990). Alternatively, the definition of a hydric soil incorporates wetland vegetation in addition to soil taxonomy. The US Soil Conservation Service defined a hydric soil as *"a soil that in its undrained condition is saturated, flooded or ponded long enough during the growing season to develop anaerobic conditions that favor the growth and regeneration of hydrophytic vegetation"*. Excluded soils are those that: a) were once wet but have been drained and b) are not naturally wet but periodically flooded or saturated for specific management purposes (Parker et al. 1985).

Restoration of a wetland can range from establishing new habitats in the wetland, to re-establishing a whole wetland, to the creation of wetland buffer zones for improving the quality of water entering the target wetland. In all cases, selection of the proper soil is a critical step for successful restoration of the wetland since it will define the number and degree of performance of wetland functions that will be established in the area. Soil properties should not only be considered when locating an appropriate wetland restoration site, but also during design and implementation of the wetland restoration plan. Some properties are applicable to both site selection and design. Within a watershed there are several soil types with different properties that are able to support numerous soil processes. Although the properties of the soil substrate set the boundaries within which the wetland ecosystem is expected to function, soil processes will finally define the degree to which wetland functions will be performed and will determine the final outcome of restoration.

The objective of this chapter is to underline the importance of soil quality in site selection for wetland restoration, to outline the role of soil properties in restoration design and to point out the effect of soil processes on the degree of performance of the restored functions. Discussion will be limited to selected important soil processes and properties, their role in the functioning of wetland ecosystems, and their application to the restoration of wetlands.

Soils and site selection for wetland restoration

Wetland functions depend on the characteristics of the structural components of the wetland. Site selection and the design of the wetland will determine the degree of performance of each wetland function. Soil is one of the structural components of wetlands that control wetland establishment. Soil selection is of critical importance for the successful restoration of wetlands. Sometimes, instead of using naturally occurring soils, sand and gravel can be used as substrate, to support vegetation and to provide the appropriate redox regime, especially in constructed wetlands. The properties of soils, or the substrate, are reflected in the functions of the wetland. Therefore, according to the desired functions of the wetland, there is a need to select the proper soil. Within a watershed, there are several types of soils. Soil quality changes in response to use and management and is characterized by properties that both range within limits and interrelate functionally to each other (Larson and Pierce 1994).

The soil processes have direct and indirect effects on the performance of the wetland to be created or restored. Regulating and partitioning water and solute flow depend on texture, structure, organic matter and mineralogy of the soil (Misopolinos and Zalidis 2000). A soil having high permeability cannot be selected for establishment of a wetland because it will be impossible to maintain proper hydrology. Furthermore, for wetlands that will receive high amounts of pollutants, the soil or the substrate should function to deposit, store and transform inorganic and organic materials.

Soil quality, as defined by soil processes, differs within and between watersheds. The major objective of soil quality assessment is to predict, from a knowledge of soil properties, the ability of the soil to support specific quality/health functions for crop, animals-humans, and water target systems (Harris and Bezdicek 1994). Because of the latter, site selection for the establishment of a wetland should consider soil quality as one of the major environmental constraints for restoration of a sustainable ecosystem that is intended to perform specific wetland functions. Soil heterogeneity within a watershed, makes site selection rather difficult. For this reason, Zalidis et al. (2002) developed a relatively easy and low cost procedure for evaluating soil quality at the watershed level. This procedure identifies areas of specific functional interest where specific management measures can be applied for uses such as crop production, construction of buildings, and wetland establishment.

Soil properties and design of wetland restoration

In their structural role, soils can function to perch and hold wetland water or serve as a pervious medium that allows groundwater to move into, through, and out of a wetland. In addition, the soils and the substrate serve as a biological interface to support macro- and microinvertebrate, and microbial populations, act as a medium for plant growth, and regulate water quality.

Engineering properties include soil strength, compressibility, and permeability. These properties influence the critical soil processes governing the performance of a wetland when a force is applied to the soil. In addition, compaction and erosion characteristics of wetland soils, sedimentation, and flow of water through wetlands and dikes are influenced by these engineering soil properties. These properties influence water and sediment storage capacity, water quality and flow, and the deposition and erosion of sediments on the surface of submerged wetland soils. The strategy for determining procedures for construction or restoration, operation, and maintenance of wetlands is based on critical soil processes evaluated from the engineering properties of wetland soils. Information required to determine these properties for the final design, construction, and maintenance of restored wetlands may result from detailed site evaluation and soil surveys. The latter should include in-situ soil tests, such as cone penetration (CPT) and standard penetration (SPT), as well as laboratory tests performed on undisturbed soil samples (Hayes et al. 2000).

In addition to engineering properties, knowledge of the physical and chemical composition of local soils is essential to predict some chemical and biological processes accurately in restored wetlands. Several soil properties (soil type, redox, pH, and salt concentration) are essential for the successful restoration of wetland functions (Table 1). An extensive review for the use of soil properties in wetland restoration is provided by the US Army Corps of Engineers (Hayes et al. 2000).

Table 1. Examples of soil/substrate properties and related characteristics influencing wetland functions (based on Maltby 1992).

Function	**Examples of important soil/substrate properties and related characteristics**
Ground water recharge	Texture, permeability
Floodwater attenuation	Texture, micro-relief, water retention
Shoreline stabilization	Structure, texture, organic matter, micro-relief
Sediment/toxicant retention	Clay mineralogy, texture, pH, redox
Nutrient removal/ transformation	Organic matter, texture, Fe/Al content, microbes, pH, redox
Food web support	Texture, organic matter, pH, redox, soil chemistry
Water storage	Texture, structure, permeability

Many natural and restored wetlands have soil/sediment profiles that often include organic and mineral horizons or layers, and also display spatial heterogeneity. Flooded organic soils can often hold more water than flooded mineral soils. In spite of higher porosity, organic soils may have low (sapric) to high (fibric) hydraulic conductivities, affecting horizontal or vertical groundwater flows. Thus, organic soils may act as the main aquitard to seal a wetland and increase the hydroperiod (Kadlec and Knight 1996).

Wetland soils form in response to external and internal loading of carbon and mineral sediments to the wetland. In closed systems without inputs and outputs, new organic sediment formation is dependent on the net difference between carbon fixation by plants and the rate of carbon degradation. In wetland systems open to inflows of water, sediments may accrete through sedimentation of mineral and organic solids. In a flow-through wetland, influent sediments tend to accumulate near the front end of the system. Usually, a newly planted and flooded wetland has no plant litter, and oxygen consumption in the sediments is limited to existing low concentrations of soil organic matter. Thus, redox potential in the new wetland sediments is relatively high compared to mature wetland ecosystems. Litter accumulation results in an additional diffusional barrier between atmospheric gases and wetland sediments and exerts a very high, internally produced oxygen demand due to high rates of microbial respiration. Very low redox levels may be reached in mature wetland sediments, resulting in anaerobic conditions as chemically bound oxygen is utilized through microbial oxidation-reduction reactions (Kadlec and Knight 1996).

In flooded soils or sediments, aerobic and anaerobic microbial metabolism may occur simultaneously at different depths depending on the availability of electron acceptors, which have different affinities for electrons and will be used preferentially by microorganisms in the order indicated by thermodynamic considerations (Table 2). The relative affinity for electrons of the various electron acceptors indicates the intensity of reduction and is related to soil redox potential. Redox potential (Eh) is a measure of electron availability in chemical and biological systems. Oxidized systems accept electrons and cause Eh values to be high and positive, while highly reduced systems donate electrons and cause Eh values to be negative.

Table 2. Hydrogen and electron acceptors in energy metabolism (adapted from Sikora and Keeney 1983, Valiela 1984, and Reddy et al. 1986).

Metabolic process	Conditions of growth	H^+ or e^- acceptor	Reduced product	Energy yield*	Eh (mV)
Respiration	Aerobic	O_2	H_2O	-686	700 to 350
	Facultative anaerobic	NO_3	NO_2, N_2O, N_2	-649	350 to 200
		Mn^{4+}	Mn^{2+}	-458	200
		Fe^{3+}	Fe^{2+}	-100	100 to -100
	Anaerobic	SO_4	H_2S	-91	-100 to -200
		CO_2	CH_4		< -200
Fermentation	Facultative or obligate anaerobic	pyruvic acid	lactic acid acetaldehyde	-58	
		acetaldehyde	ethyl alcohol		

* kcal/mol glucose

Prior to flooding, soils may display pH values from 3 to 10 (Hammer 1992). Following flooding, the pH in wetland soils may initially decline due to anaerobic decomposition liberating carbon dioxide into interstitial water. This phase is generally transient and is followed by a shift in both acidic and alkaline soils toward pH neutrality over time. The pH decrease of submerged alkaline soils results from the build up of carbon dioxide, however, the pH increase in acid soils is mostly the result of Fe^{3+} oxyhydroxide reduction. In the case of a presence of contaminants, and depending on the contaminant removal mechanisms and soil type, pH changes of this magnitude could have impacts on the pollutant removal and retention functions of a wetland. Natural wetlands exhibit pH values ranging from slightly basic in alkaline fens to quite acidic in sphagnum bogs (Mitch and Gosselink 1993). Organic substances generated within a wetland are the source of natural acidity. The resulting humic substances are large, complex molecules with multiple carboxylate and phenolate groups. The protonated forms tend to be less soluble in water and precipitate under acidic conditions. As a result, wetland soil-water systems are buffered against incoming basic substances (Mitch and Gosselink 1993).

In addition to the former, salt concentration is a factor of particular importance for Mediterranean soils. Salt accumulation in Mediterranean soils is a natural process favored by the region's environmental conditions (Zalidis et al. 1998). Considerable areas of salt-affected soils have developed following large-scale flood control and wetland drainage projects. Presently, most salt accumulation is due to deterioration of the quality of the groundwater used for irrigation. This deterioration was caused by overpumping and the associated intrusion of sea water. Wetland restoration in such cases is of a great importance for the remediation of both the soil and water resources in the area. For the restoration of a freshwater wetland in salt-affected areas, wetland water should be continually replenished by low EC water from the watershed to suppress salinity in the ecosystem. If water is replenished at a suitable rate, it will be possible to keep salt concentrations down to acceptable levels. To achieve this aim, it will be necessary when working out the water balance, to allow for loss of water used to leach out salts (Zalidis et al. 1998).

Restored wetland functions as a result of soil processes

Many of the changes that occur during the succession of a restored wetland ecosystem are the result of biological factors such as the growth of bacteria and fungi, algae and macrophytes, micro- and macroinvertebrates, and larger animals that occur in wetlands. While many of these natural processes are not within the control of the wetland designer, their effects should be considered when trying to maximize chances for success in a wetland restoration project. Properties of the wetland soil by themselves cannot determine the final state of the restored ecosystem. In order to predict the final outcome of restoration, the designer should also take into account the soil-related processes and their effect on the biochemistry of the wetland. Soil processes regulate the capacity of a soil to function within ecosystem boundaries to sustain biological productivity and environmental quality and promote plant and animal health (Doran and Parkin 1994). Soils serve as sinks, sources or transformers of carbon, nutrients, and chemical contaminants, and thus determine the degree to which wetland functions will be restored. Therefore, designers are cautioned to consider not only the results of the standard soil tests, developed mainly for aerobic systems, but also a number of key soil processes that take place after restoration.

The relation of water saturation with soil reduction or anaerobiosis gives wetland soil substrates, including sediments, unifying properties in terms of biochemical processes taking place within them and brings forth a generalized multi-zone model. When soils are flooded and saturated, oxygen is consumed, by biological and chemical processes (Gambrell and Patrick 1978) faster than it is supplied by diffusion in the liquid phase. Thus, it is generally assumed that oxygen in the soil disappears within hours. Under these conditions, several anaerobic zones are formed below a thin aerated or oxidized zone as the soil becomes progressively reduced with depth. This type of vertical arrangement presents a model for explaining biochemical transformations occurring in wetlands. Although it is typical of situations where shallow flooding of soils occurs, it is sometimes overly simplistic because saturation does not always result in anaerobic soil conditions. For example, oxygen presence in deeper soil can be attributed to lack of organic carbon, which would otherwise serve as an energy source (electron donor) for microbial metabolism (Gambrell et al. 1991). Dissolved oxygen is also affected by the permeability of the soil as determined by the size of pore spaces, which are in turn, a function of bulk density and soil texture. Thus soils or parts of soil profiles with low clay content are less likely to develop anaerobic conditions (Stolzy and Fluhler 1978).

Reduced soils and sediments are effective sinks for most metal contaminants and a number of organic pesticides. Toxic metal immobilization is also of interest when considering sludge disposal in such systems. The uptake of Cd and Zn by rice and corn decreases as sludge-amended soil becomes more reduced (Gambrell 1994). Increased metal retention capacity by complexation (especially for Cd and Zn, but also for Pb, Hg, Co) in neutral anaerobic systems is not caused by the direct effect of low redox potential, but by the presence of more humic material, which is of higher molecular weight and of greater structural complexity (Gambrell and Patrick 1978, Reddy et al. 1986, Gambrell et al. 1991). Highly insoluble metal sulfides are a major metal-immobilizing process in coastal sediments (Gambrell 1994). Other metal oxides, hydroxides and carbonates of low solubility also have been implicated as forming stable metal complexes. However, metals immobilized in such forms are potentially available, and transformations affecting their mobility and availability can take place by means of changes in the physico-chemical properties of the receiving soil substrates. These properties include pH, redox potential and salinity (Gambrell and Patrick 1988). The relative importance of soluble organic acids in maintaining metals both in solution and in an available form is unknown.

Plant litter is the main source of organic matter input in wetlands, whether produced in situ (autochthonous origin) or transported from adjacent ecosystems (allochthonous origin). Organic detritus ultimately enters the soil through precipitation and sedimentation, and soils become the major sites of organic matter transformations. Such transformations occur through microbial decomposition, although chemical oxidation may be important at high temperatures. Carbon is used in microbial metabolism directly for cell growth and indirectly for the production of energy.

The type of microbial metabolism and the nature of the transformed carbon products are determined by the degree of soil aeration, which, in turn, controls the depth of the surface aerated zone and the residence time of organic detritus within that zone. In flooded soils, decomposition of organic matter takes place predominantly in the underlying anaerobic zones. Anaerobic microbial metabolism involves a number of inorganic electron acceptors such that organic matter decomposition is a dominant mechanism for carbon and nutrient transformations (D' Angelo and Reddy 1994). However, restored wetlands frequently have lower soil organic carbon and microbial activity relative to natural wetlands. The establishment of a viable microbial component will enhance the capacity of the system to transform organic matter and nutrients, and it has been one of the most elusive goals in wetland restoration.

The principal reserve of plant nutrients in soils, although not always true for P, is the organic fraction. The amount of inorganic N and P, but not always that of S, in soil solution is directly related to the amount of utilizable organic matter (Reddy et al. 1986). The availability of released inorganic forms will be induced by changes in redox potential. For example, in reduced soils and sediments, decomposition of organic N will yield NH_4^+ for the support of vegetation through plant uptake due to the low N requirements of anaerobic bacteria. The diffusion of excess NH_4^+ upward to the thin surficial oxidized zone through a concentration gradient results in the formation of nitrates by nitrification. Nitrates will diffuse downward to the anaerobic zone through a concentration gradient, and N_2 loss through denitrification will occur. This nitrification-denitrification sequence explains the loss of 20-50% of applied N to flooded soils (Gambrell and Patrick 1978). Although the loss of nitrate to gaseous products is not beneficial with respect to N economy in agricultural soils, dentrification losses may be desirable for preventing ground and surface water pollution; swamp and marsh soil substrates have a large capacity to remove NO_3-N.

Organic phosphorus comprises a substantial reservoir in reduced soils and sediments, but its slow mineralization rate has directed attention to inorganic forms of P (Gambrell and Patrick 1978). The accumulation of organic matter in soil creates highly reduced conditions, which solubilize oxidized P-containing forms of Fe and enhance P availability (Gambrell and Patrick 1978, Gale et al. 1994). The thin, oxidized surface zone acts as a barrier to P diffusion upward because oxidation leads to the formation of relatively insoluble forms, i.e., ferric phosphate and P-occluded in ferric oxy-hydroxides (Gambrell and Patrick 1978, Gambrell et al. 1991).

Under anaerobic conditions, organic S is reduced to odiferous mercaptans and sulphate to sulfide by anaerobic bacteria. Plant roots oxidize sulphides by oxygen; diffusing out of the roots, and the plants (Drew and Lynch 1980) can take up the sulfate. Micronutrient availability is also affected by oxidation-reduction and pH values generally rise in flooded wetland soils. Reduced Mn and Fe solubilize and result in greater plant availability in acid soils. K, Ca and Mg are also found in greater concentrations in soil solution under reducing conditions (Reddy et al. 1986).

Conclusions

The selection of the proper soil (or other kind of substrate) for the restoration of a wetland within a watershed area is a challenging task for wetland scientists. Since there is a wide range of soil types with variable texture, structure and organic matter content within each watershed, soil quality becomes one of the critical site selection criteria that should be considered for the restoration of wetlands.

Water saturation is the common soil feature that is the basis for the development of unified concepts in soil taxonomy and the biogeochemistry of wetlands. Soil microbial populations have significant influence on the chemistry of most wetland soils. Important transformations of nitrogen, iron, sulfur, and carbon result from microbial processes. Water saturation, in combination with the large inputs of organic matter, promote the development of anaerobic conditions, which alter and delay the biochemical process of decomposition compared to that of drained upland soils. As a result, the soil properties and the related processes, in the area where the wetland will be re-established, should be of major concern in order to predict and ensure the sustainability of the wetland functions that will be restored.

The biochemical processes of the wetland soil play a critical role in the successful application of the emerging technology of restoration. Wetland and soil scientists need to define common variables among these individual conditions, to identify mechanisms responsible for establishing wetland functions, and to incorporate these into scientifically sound criteria for the restoration of wetland ecosystems.

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RESTORATION OF AQUATIC VEGETATION IN MEDITERRANEAN WETLANDS

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Introduction

Wetland vegetation is the structural and functional backbone of the wetland ecosystem and largely determines the habitats of other wetland organisms (Verhoeven 1992). Plant communities are key components of the biological diversity of wetlands. They can be used as indicators of wetland functions, because aquatic plants provide the energy support for the rest of the trophic web. The botanic landscapes form an outstanding asset for recreation in many wetlands. Although nutrient availability and vegetation management play an important role in vegetation composition, hydrology is the governing factor that determines species richness and vegetation extent. Water level, duration and seasonality of flooding, and substrate saturation has a strong influence on the composition, stability, distribution, and performance of wetland vegetation (Wierda et al. 1997).

Wetland flora respond even to slight changes in hydrological conditions with changes in species richness, diversity, and productivity (Mitsch and Gosselink 1993). Observed responses of wetland plants to hydrological changes depend on the species and on the magnitude of changes. Such responses may include a reduced distribution range, or complete disappearance of species incapable to adapt to a new environment. Some species persist in sub-optimal conditions but with reduced vigor. Others may show even an enhanced growth in the new water depth range. The new habitats may be colonized by a range of new species (Wheeler et al. 1995).

In addition to alterations of the water regime, the flora and vegetation of Mediterranean wetlands are subjected to the effects of harvesting, burning, anthropogenic inputs of nutrients and toxicants, the presence of exotic species etc. which may also cause alterations. There are cases where restoration of plant communities is possible by simply arresting the causes of alteration. There are other cases where additional measures are necessary, such as artificial re-establishment of the wetland plants.

The first task in the restoration of wetland ecosystems is, of course, to secure the necessary water resources so as to reinstate the water regime to the highest degree possible. Also, to secure appropriate substrate for the rooted plants. The second is to ensure the recovery (or the re-establishment when necessary) of target plant communities. The information needed to meet the second task pertains mostly to wetlands of North America and North and Central Europe. The plant communities of Mediterranean wetlands have been relatively little studied and there are few examples of restoration projects.

This chapter briefly presents the common plant communities in Mediterranean wetlands and the causes of their alteration. The approaches to address these causes and the factors to be considered towards reaching recovery and/or successful re-establishment are discussed. The importance of setting clear and feasible restoration targets and selecting suitable techniques to evaluate the achievement of these targets are pointed out. Finally, two examples of vegetation restoration projects from Israel are presented.

Common plant communities in Mediterranean wetlands

In this chapter the CORINE classification system of wetlands (Table 1) is used in order to avoid confusion in terminology. This system is applied in all Mediterranean countries of the European Union (EU) as well as in candidate countries (Commission of European Communities 1991). Information on the present distribution of wetland vegetation in the Mediterranean basin comes from the ETC/NBC (2000) database (Commission of European Communities 1992-Directive 92/43/EEC), and from other sources (e.g. Dafni and Agami 1976, Hollis 1986, Britton and Crivelli 1993, Rívas-Martínez et al. 1993, Grillas and Roche 1997, Papastergiadou et al. 1997).

The most important habitat types[1] which can be distinguished in Mediterranean wetlands according to available information, are presented in Table 1. The code numbers are common for all EU countries and included in the Natura 2000 database. The asterisk * means "priority habitat type" needing urgent conservation measures. The geographical distribution of the above habitat types in EU and in non EU Mediterranean countries, based on available bibliography, is also given in Table 1.

[1]According to Directive 92/43/EEC, the term habitat type may correspond to several types of plant communities. Most commonly it corresponds to the syntaxonomic level of "alliance".

Table 1. Wetland habitat types based on CORINE classification in the Mediterranean. *The asterisk indicates priority habitat type. Habitat types coding, according to Directive 92/43/EEC.*

1. Coastal and halophytic habitats

1130	Estuaries. Zosteretea or vegetation of brackish water *Ruppia maritima, Spartina, Sarcocornia* etc.
1150*	Lagoons with or without vegetation from *Ruppietea maritimae, Potametea, Zosteretea, Charetea*. At the coasts of the European Union, especially in the Mediterranean basin.
1310	*Salicornia* and other annuals colonizing mud and sand (Thero- Salicornietea). Widespread distribution throughout the Mediterranean basin.
1320	*Spartina* swards (*Spartinion maritimae*), distributed to the coasts of Italy, Spain and Portugal.
1340*	Inland salt meadows; identified only from Italy.
1410	Mediterranean salt meadows (*Juncetalia maritimi*). Common in Mediterranean countries.
1420	Mediterranean and Thermo-Atlantic halophilous scrubs (Sarcocornetea fruticosi). Well distributed in the Mediterranean basin: Tunisia, Cyprus, Israel etc.
1430	Halo-nitrophyllous scrubs (Pegano-Salsoletea), characteristic of the Eastern Iberian Peninsula, present also along the coasts of France and Italy.
1510*	Mediterranean salt steppes (Limonietalia), along Mediterranean coasts and the fringes of Iberian salt basins. It is a priority habitat type needing emergency conservation measures throughout the Mediterranean. In Greece, it is recorded only at few sites (3) and it is well distributed only in Spain.

2. Freshwater habitats

3120	Oligotrophic waters containing very few minerals generally on sandy soils of the Western Mediterranean with *Isoetes* sp. It is reported from France, Italy, and Spain.
3130	Oligotrophic to mesotrophic standing waters with vegetation of the *Littorelletea* uniflorae and/or Isoeto-Nanojuncetea. Very rare in Greece, but well distributed in other Mediterranean countries; needs further investigation.

Table 1. Wetland habitat types based on CORINE classification in the Mediterranean (continued).

3140	Hard oligo-mesotrophic waters containing benthic vegetation of *Chara* sp. Not well studied.
3150	Natural eutrophic lakes with Magnopotamion or Hydrocharition-type vegetation. It is well distributed in the Mediterranean basin.
3160	Natural lakes and ponds with brown tinted water due to peat and humic acids. Plant communities belong to the Utricularietalia order. It has a sporadic appearance in the Mediterranean.
3170*	Temporary ponds. Exist only in winter or spring with flora mainly composed of Mediterranean therophytic and geophytic species belonging to the alliances: Isoetion, Nanocyperion flavescentis, Preslion, Agrostion, Helochloion and Lythrion tribracteati. It is one of the most threatened habitat types throughout the Mediterranean basin due to human disturbances, especially hydrological.
	Reed beds (*Phragmites australis*, *Scirpus lacustris*, *Typha* sp.) and large sedges (*Carex* or *Cyperus* sp.). Not included to Directive 92/43/EEC. It is well distributed throughout the Mediterranean.
3250	Constantly flowing Mediterranean rivers with *Glaucium flavum*. In Greece it is noted only from Crete and Mount Athos. It is very sporadic and degraded throughout the Mediterranean.
3260	Water courses of plain to montane levels with the *Ranunculion fluitantis* and *Callitricho-Batrachion* vegetation.
3270	Rivers with muddy banks with *Chenopodion rubri* p.p. and *Bidention* p.p. vegetation. Sporadic in Spain, France, and Italy.
3280	Constantly flowing Mediterranean rivers with Paspalo-Agrostidion species and hanging curtains of *Salix* and *Populus alba*. It is well distributed in EU Mediterranean countries and also referred from Israel.
3290	Intermittently flowing Mediterranean rivers of Paspalo-Agrostidion. It is referred only from some places of Greece, Italy, France, and Cyprus.

3. Semi-natural tall-herb humid meadows

6410	*Molinia* meadows on calcareous, peaty or clayey-silt-laden soils (Molinion caeruleae). Very restricted to the Mediterranean, not well studied.
6420	Mediterranean humid grasslands of tall grasses (Molinio-Holoschoenion) and rushes, widespread in the entire Mediterranean basin, extending along the coasts of the Black Sea. Also reported from Cyprus.

Table 1. Wetland habitat types based on CORINE classification in the Mediterranean (continued).

6430	Hydrophilous tall herb fringe communities of plains and of montane to alpine levels.

4. Raised Bogs and Mires and Fens

7110/ 7130*	Active raised bogs/Blanket bogs. Typically present in Northern EU countries. A few relics are in Mediterranean countries needing further investigation.
7140	Transition mires and quaking bogs. These belong to the Caricetalia fuscae. It is referred only from a few sites in Greece and Italy but needs further investigation.
7210*	Calcareous fens with *Cladium mariscus* and species of the Caricion davallianae. Very restricted in the Mediterranean basin due to the expansion of agriculture and drastic hydrologic disturbances.
7220*	Petrifying springs with tufa formation (Cratoneurion). Referred from Spain, France, and Italy, but needs further investigation.
7230	Alkaline fens. They are among the habitats that have undergone the most serious decline. Endangered.

5. Riverine forests

91E0*	Alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior*. Greece, Spain, Italy, Cyprus.
92A0	*Salix alba* and *Populus alba* galleries. It is well distributed in Greece, France, Italy, and Spain.
92B0	Riparian formations on intermittent Mediterranean water courses with *Rhododendron ponticum*, *Salix* and others. It is reported only from Spain and Portugal.
92C0	*Platanus orientalis* and *Liquidambar orientalis* woods. *Platanus orientalis* gallery forests of Greek and Southern Balkan watercourses. Also reported from Cyprus and Israel.
92D0	Southern riparian galleries and thickets (Nerio-Tamaricetea and Securinegion tinctoriae). It is well distributed throughout the Mediterranean including Cyprus and Israel.

The main factors that determine the species composition of plant communities in the wetlands of the Mediterranean region are hydroperiod, depth of flooding, periodicity of flooding, water and soil salinity, grazing intensity, strength of water currents, water quality, soil or sediment type and annual temperature range. Detailed description of wetland vegetation and the ecological characteristics of plant communities is lacking except for a few published works mainly from the Western Mediterranean EU countries.

In Mediterranean wetlands, halophyte communities are generally species poor, with cosmopolitan distribution (Waisel 1972). The perennial salt marsh community of *Arthocnemum fruticosum* has characteristic species whose center of distribution lies within the Mediterranean basin. Also, *Juncus maritimus* is a Mediterranean species characteristic of the oligohaline and mesohaline wetlands and has its upper limit of tidal salt marshes along the Atlantic coast (Braun-Blanquet 1951). In deltas, *Tamarix* stands occupy large areas. Various species of this community with similar structure occur throughout the Mediterranean, for example *Tamarix africana* and *T. canariensis* are mainly distributed in Spain and Portugal, *T. gallica* in France, and *T. tetrandra* in the Balkans. Currently, this type of vegetation suffers from human interventions, such as, overgrazing, burning during summer, the use for recreation and expansion of cultivated fields.

Phragmites reed beds are widespread because *P. australis* is the most competitive species and it occurs in a wide range of salinity and flooding conditions. It often forms dense monospecific stands, which shade out almost all other plant species. Depending on the specific ecotype, *P. australis* tolerates salinities of up to 0.2 M NaCl. At higher salinities, it is replaced by *Scirpus maritimus*. In nutrient-poor wetlands, communities of *Cladium mariscus* in frequently flooded marshes, or *Molinia caerulea* in drier conditions, replace *P. australis*. In the absence of grazing, succession develops towards communities dominated by *Salix* sp. or by *Alnus* sp. Under light grazing, *Phragmites* is replaced by wet meadow communities dominated by *Cladium mariscus* or *Scirpus* and *Carex* species. Reed beds dominated by *Phragmites* occur throughout the Mediterranean region but are scarce in N. Africa, where they are limited by salinity, grazing, and drought (Britton and Crivelli 1993). Wetland drainage and increasing exploitation, have greatly reduced the area of wetland vegetation (Duhautois 1984).

Saw-sedge marshes dominated by *Cladium mariscus* can form dense, almost monospecific, stands in permanently wet areas. In Mediterranean wetlands, however, such vegetation is extremely localized, and it is not entirely clear which environmental factors favor its dominance over the common *Phragmites*, *Scirpus*, and *Typha* species. Although protected as a priority habitat, there are very few surveys and studies conducted and there is no complete inventory, except the recent database for the Natura 2000 network in EU countries. The largest known Mediterranean *Cladium* marshes-peatlands are in the Tablas de Daimiel in Central Spain, the Marais de la Crain Southern France, around some lakes on the Tyrrhenian coast of Italy, the historic Philippi fen in Eastern Macedonia (Greece) and in many smaller areas which are often damaged. Low nutrient status is thought to be a factor favoring the dominance of *Cladium* over *Phragmites* in Northern Europe (Wheeler 1980).

Another formerly widespread vegetation type in Mediterranean wetlands is wet meadows. This is a heterogeneous assemblage of plant species occurring in areas that are flooded during winter and keep a high groundwater table in summer. Occasional burning, mowing, or grazing is applied which prevents scrub invasion and discourages the growth of *Phragmites australis*. Wet meadows are widespread in Northern Europe but restricted in the Mediterranean, where summer drought limits the growth of many characteristic species. They often occupy an intermediate position between reed beds and terrestrial vegetation. *Butomus umbellatus, Eleocharis palustris, Iris pseudacorus, Juncus gerardii, Alisma* sp. are the most abundant species. In the coasts of Israel, those plants characterize isolated ponds between sand dunes. Stands of *Molinia caerulea* form another wet meadow association on the margins of grazed *Cladium* fens along the coasts of Spain, Algeria, and Northern Greece (Britton and Crivelli 1993). *Molinia* stands are flooded to about 20 cm from October to March-April but dry in summer (Moubayed 1977). In drier places the plant communities are dominated by *Carex* sp. and *Scirpus holoschoenus*.

In freshwater or slightly saline wetlands, the submerged vegetation is species poor and tends to be monospecific. The most widespread species are *Potamogeton pectinatus* and *Myriophyllum spicatum* (Papastergiadou 1990). In calcareous waters, characteristic species are *Chara vulgaris, C. aspera, Nitella flexilis, Potamogeton pusillus, Callitriche* sp., *Zanichellia* sp. Rooted floating-leaved associations are uncommon in the Mediterranean region, due to large annual fluctuations in water level and wave action caused by strong winds. An association of endangered species *Trapa natans* and *Salvinia natans* is found particularly in Northern Greece (Lake Ismaris, Lake Kerkini and Evros river), where it covers large areas of water.

Riverine forests occur along flowing water courses, on adjacent floodplains, and these areas are flooded by winter rainfall or spring snow melt and the groundwater table is high for much of the summer. The characteristic association is the Populetum albae, a community restricted to the Mediterranean region, with characteristic species *Populus alba, Alnus glutinosa, Ulmus* sp., *Salix alba* in wetter places, *Amorpha fruticosa* a species introduced from Bulgaria etc. Another very common plant association in Greece and other Balkan countries is the Platanetum orientalis, which is dominated by *Platanus orientalis*. The riverine forests in the Mediterranean region are highly fragmented. The largest remnants occur in the Po river valley and along the lower Rhône, but small vestiges are scattered along all the major rivers in Southern Europe. In North Africa riverine forests are very scarce (Britton and Crivelli 1993).

The best known exotic species in the Mediterranean wetlands are listed in Table 2.

Table 2. Indicative lists of exotic, endangered, and threatened plant species in Mediterranean wetlands.

Exotic species
Azolla filiculloides: neotropical species introduced in Europe as feed in aquaculture. It has expanded very fast throughout small rivers and ponds and it is very antagonistic to *Lemna* species.
Lythrum salicaria: introduced from Central America and today distributed all over Europe.
Paspalum paspalodes, P. distichum: a grass species introduced from Central America to the Mediterranean with rice seed.
Ludwigia grandiflora, L. peplodes: introduced from sub-tropical America. They have been colonized France since the 19th century.
Eichornia grassipes: introduced from the tropical regions into Europe.
Amorpha fruticosa: introduced from Bulgaria to Northern Greece (river Strymon).
Endangered and threatened species
Damasonium stellatum: one of the rare and threatened species of temporary marshes.
Marsilea quadrifolia and Marsilea strigosa: reported only from one or a few sites throughout Mediterranean.
Trapa natans: endangered.
Salvinia natans: endangered.
Riccia fluitans: endangered.
Ricciocarpus natans: endangered.
Nuphar lutea: endangered.
Cladium mariscus: threatened.
Cyperus papyrus: endangered.

Causes of alteration of wetland plant communities

The conditions under which the wetland plant communities occur in a given site are determined by the topographical and hydrological patterns of that site (Tallis 1983). However, during the last 100 years human-induced changes in wetland vegetation have drastically increased. Many Mediterranean wetlands have been highly affected by human interventions resulting in the alteration of typical species composition and vegetation structure, and in the formation of species-poor plant communities and, finally, in substrate degradation. The most serious interventions are:

Changes in the water regime

Draining a wetland ecosystem means lowering the water table and drying out the surface layers. Colonization follows drainage and represents an acceleration of the normal pathways of the hydroseral change. Marginal wetland communities are also affected. In the Mediterranean countries the most common reasons for draining wetlands were to combat flood damage to property, to acquire new farmland, and to control mosquitoes.

One ecological goal for many restoration projects in lakes which have become shallow and are covered by vegetation, is to provide the appropriate water regime in order to create an open water area together with a mosaic of open water, submerged and emergent plant communities.

Burning

Deliberate burning of wetland vegetation is practiced for various reasons. If burning occurs when the surface layer is wet, then only the removal of the aboveground vegetation may result, and natural revegetation from the undamaged underground plant organs quickly follows. Under dry conditions, or when the fire is particularly fierce, oxidation of the surface organic layers may occur, with a consequent annihilation of many plants. Burning also affects a wetland area in a non-uniform manner, because of local variations in the wetness of surface layers. After severe fires the initial recolonization stage is dominated by "nitrophilous" or "halophilous" species, such as *Urtica dioica*, *Aster tripolium*, or *Solidago* sp. (Vogl 1973). Periodic burning is widely practiced in reed bed vegetation stands in Mediterranean and even in Ramsar wetlands (e.g. Lake Prespa, Lake Ismaris, Evros Delta), in order to promote new plant growth for grazing animals. This affects the composition of adjacent plant communities. Thus, regular burning may lead to the temporary stability of wetland communities, but with the probability of long-term degradation of the ecosystem through nutrient losses and soil erosion.

Direct or indirect harvesting of vegetation

Mowing or grazing practices in wetland plant communities introduce another major environmental factor to the system. Plant communities of wet meadows, salt marshes, reed beds, or sedges, are regularly utilized for the acquisition of hay or thatching materials. Mowing is mostly carried out on semi-terrestrial plant communities and can lead to the suppression or even elimination of native plants and to their replacement by species that are better adapted to regular mowing (Tallis 1983).

Input of plant nutrients

Waters flowing into a wetland unavoidably carry plant nutrients (e.g. nitrogen, phosphorus) dissolved in the water and adsorbed in soil particles. Depending on the watershed these nutrients may originate from agricultural, municipal or, industrial sources. The inputs from these sources are often many times greater that the inputs from natural sources, i.e., from the areas of the watershed which are covered by natural vegetation. One effect is to accelerate eutrophication.

Eutrophication is a principal cause for concern for many lowland water bodies. It is often blamed for aquatic plant problems, at least during the early stages of nutrient enrichment (Vollenweider 1971, Agami 1984, Giulizzoni et al. 1984, Murphy 1988). Advanced eutrophication is commonly associated with macrophyte diebacks and replacement of macrophytes with periphytic and planktonic algae (Lachavanne 1985, Hosper 1989, Moss et al. 1997). The lowering of nutrient levels is frequently advocated as a long-term strategy for the management of nutrient-enriched water bodies. Although rises in nutrient levels enhance fish production, the loss of habitats (e.g., by sediment build up, deoxygenation, undesirable proliferation of macrophytes) and the loss of food sources (by food web simplification) cause a shift from desirable game fish to less desirable species, especially in extreme cases of eutrophication. Many known examples are derived from shallow lakes in Greece such as Lake Kastoria, Lake Pamvotis, and Lake Cheimaditis.

Many highly eutrophic lakes develop increasing amounts of emergent littoral vegetation and allow terrestrial vegetation to encroach (Wetzel 1983). A common succession of shallow wetlands follows this pathway:

Macrophytic vegetation and associated microflora → Reed swamps → Marshes and fens → { Moss vegetation → Bog systems; Terrestrial vegetation }

Dominance of one species or co-dominance of only a few species in wetlands principally means that the system is extreme in some way because of some limiting factors. In such systems vegetation is structurally underdeveloped.

Reed swamps are characterized by dense vegetation of the littoral species (usually of tall grass species such as *Phragmites*, *Cyperus*, *Scirpus*, and tall *Carex* species) with only occasional mixtures of submerged and floating leaved macrophytes. As organic matter accumulates and displaces the standing water, the water-logged organic substrate is often characterized by moderate-sized graminoid species and small herbs stratified in two or three layers. Such type of vegetation is referred to as "marsh" or "fen" in areas of very slowly flowing surface water, or "bog" in areas where deep seepage occurs (Wetzel 1983).

Input of toxicants

The presence of toxic substances is reported for Mediterranean wetlands. Such substances mainly include pesticides (used directly on the wetland to control hydrophytes and mosquitoes or used in the adjacent farmland to control crop pests), heavy metals, radioactive materials, and various chemicals from industrial plants which discharge their untreated effluents into wetlands. Direct toxicity occurrences in wetland vegetation are seldom reported. The most widely known accidental introduction of toxicants in a Mediterranean wetland is the case of the Doñana wetland in Spain. Another example is Lake Koronia, a Ramsar wetland in Greece that received toxicants from its watershed for many years. Toxic metals were found in the food web of the lake (Bobori 1996).

The presence of pesticide residues (Albanis 1992) and heavy metals (Zalidis et al. 1999) in the Axios Delta has been documented, but the effects on vegetation were not studied.

Presence of exotic species

Table 2 lists the exotic species found in Mediterranean wetlands. The magnitude of the problem is not well documented. Most of these species were not deliberately introduced. There is, however, a major case of deliberate introductions, namely, the replacement of natural riverine forests by poplar plantations. In most of those plantations allochthonous genetic material was used. Invasions of aquatic weeds causing major management problems all over the world has been reported (Hutchinson 1975, Barret 1989).

Determination of restoration targets for plant communities

There are several aspects to be considered in planning and target setting for the restoration of a whole wetland ecosystem (see also other parts in this book). With specific regard to the vegetation component of the ecosystem, its restoration targets must be such as to serve the general targets of the ecosystem. For example, if bird diversity is the principal target for the restored wetland then its venation should have the most appropriate plant species composition and architectural structure, which the site may sustain in order to provide shelter, food, rest and reproduction opportunities for the highest number of bird species.

The concept of target communities seems a very interesting one. For example in the Netherlands they developed an entire system of target communities for the whole country. Taking into account data bases of releve's made in different periods, including their coordinates could reveal which plant communities have occurred in the past in which parts of the country. Hence adapting the target community concept to regional levels and taking into account historical data would render a more realistic fit of target communities to be aimed at in ecological restoration (Bakker et al. 1998). Target communities often include many Red List species. Many of them are so strongly endangered, and their ecological requirements are rarely fully understood, that it is a difficult and very ambitious task to get them back.

To define the target plant community for a particular site it is important to collect quantitative information on: a) the composition and the structure of communities existing within the site, b) the environmental conditions of the site (climate, substrate, water regime), c) past and present management practices in the site and its watershed.

There may be conflicts in deciding whether to strive for increased diversity overall or to promote conditions favorable for a particular rare species. Also, great structural diversity often leads to increased species diversity. Thus, a lake with fringing reed-swamp, floating and submerged plants is highly desirable from a nature conservation viewpoint as it benefits all other groups of organisms. Variation in physical features (depth, profile, and sediment type) is also desirable.

Recovery of plant communities in an altered wetland

There are several cases whereby plant communities even in severely altered wetlands may start recovering their species composition structure and productivity in variable periods and degrees after the cause of alteration is arrested.

The most powerful means to achieve recovery is to bring back the water flow (in wetlands which had shown an appreciable water flow), water level, and periodicity of flooding to their former status. Some communities, e.g. those dominated by *Cladium mariscus*, are favored by flow while others, e.g. those dominated by *Carex elata*, are favored by stagnant waters.

Several Mediterranean fresh water wetlands have become much shallower during the last decades due to excessive pumping of their water for irrigation. A notable example is Lake Cheimaditis in Western Macedonia, Greece. This resulted in an extension of the reed beds (dominated by *Phragnites australis*) over most of the wetland area, which led to partial loss of the biodiversity and fishing values. Clearly, one way to restrict the reed bed zone to what it was 50 years ago is to raise the water level by restricting the extraction of water out of this lake and facilitating the flow of water into the lake. One expects that the increase of open water areas, especially if coupled with an increase in water transparency, will favor several species of aquatic flora, which are presently rare or absent.

Mediterranean wetland plant communities have evolved with man's presence as a harvester of wetland plants and as a livestock farmer for thousands of years (Papanastasis 1992). Reeds were harvested for thatching, basket weaving, animal feed etc. or sometimes burned in years of drought to provide more farmland or rangeland. They were also grazed by water buffaloes and less so by cows. Wet meadows were mowed for animal feed or directly grazed. Wood was selectively cut from riverine forests for house construction and fuel.

These practices started to change in most wetlands, especially in the north Mediterranean, during the 1950's as follows: a) **Reeds**: They were seldom cut and removed since their traditional uses were abandoned. Free grazing water buffaloes disappeared from many countries. For example, in Greece, small herds are now found only in three wetlands (Georgoudis et al. 1994). Reed bed burning is still occasionally practiced although in Ramsar sites it is generally forbidden. b) **Wet meadows**: They have become an easy prey to agricultural development (drainage and cultivation). The few areas that escaped drainage are in some cases overgrazed (e.g. Evros Delta, Axios Delta), while in others (e.g. Lake Mikri Prespa) grazing has almost stopped for socioeconomic reasons and woody plants started to appear. c) **Riverine forests**: Most were clear-cut and turned into cropland. Some of those remaining are threatened by unsuitable management of the river water level, which prevents their natural regeneration (e.g. Strymon river, Lake Kerkini), and others by river embankment, which prevents their seasonal flooding and the consequent enrichment of the soil with nutrients. This enrichment has for ages been the primary cause for their very formation and maintenance because their soils are inherently nutrient poor.

Reinstating the appropriate water regime will bring recovery. However, in several cases additional measures are necessary. Reed cutting and removal should be practiced in many shallow lakes to produce diverse plant and therefore animal communities (Husak 1978) and to slow accelerated eutrophication. This practice, however, is constrained by high cost and lack of data. Cost may be lowered by using the same harvesting machine for several neighboring wetlands. A cost-free alternative to mechanical cutting may prove to be reed grazing by water buffaloes. This magnificent animal can graze in much deeper waters than any other farm animal. Its re-introduction to a wetland will also enrich the landscape's aesthetic value. An ongoing experimental re-introduction of a very small herd of water buffaloes in Prespa National Park, Region of Western Macedonia, Greece, has yielded encouraging results (I. Kazoglou 2001, personal communication). Long-term experiments are needed to answer questions such as when, how often, and what percentage of the reed bed area of a wetland to cut.

Overgrazed wet meadows may easily recover by preparing and enforcing sustainable range management plans. In undergrazed ones, sustainable grazing by cows and water buffaloes, and not by sheep and goats must be re-introduced and woody plants must be controlled. Noteworthy studies on the sustainable management of these invaluable habitats for wildlife are under way in several Mediterranean Ramsar sites by the Station Biologique de la Tour du Valat, Arles, France.

Overflooding of riverine and lacustrine forests must be arrested and artificial regeneration techniques must be applied whenever necessary (Vitoris 2000). Unfortunately, reinstating seasonal flooding in riverine forests may prove a very difficult venture because of the increasing demands for drinking and irrigation water and the embankment, change of flow and damming of many rivers flowing into the Mediterranean sea.

Inputs of toxic substances, such as heavy metals and pesticides, have been documented as a ubiquitous phenomenon in Mediterranean wetlands. Documentation of the effects of these substances on plant communities is scarce. However, inputs of nutrients (especially phosphorus and nitrogen) from point sources (e.g. municipal wastewater) and non-point sources (e.g. cultivated fields in the watershed) are quite common and their effects on wetland autotrophs obvious.

Non-point agricultural pollution with nutrients can best be faced by adopting sustainable management practices in the agro-ecosystems of the watershed. In the 1990's the European Union provided incentives to farmers in its member states to adopt such practices. There are two factors constraining the effectiveness of these incentives: Firstly, there is a paucity of research information and secondly, insufficient training projects have been developed for farmers (A. Gerakis et al. 1997).

Wastewater treatment installations are multiplying in the North Mediterranean. Several of them fail due to ineffective planning and high operating costs. On the other hand, wetlands constructed for wastewater treatment may be a supplementary low cost approach. Such a wetland is planned to be established close to Lake Karla, Greece (a drained lake currently under restoration). Drainage waters from the perimetric cultivated zone will be first discharged into the constructed wetland for treatment and then led into the restored lake (Zalidis et al. 1995).

Wetland water quality improvement efforts may use biomanipulation techniques as a supplementary tool (Moss et al. 1997). Such an improvement implies changes in nutrient cycling. Successful biomanipulation of shallow lakes has often involved massive growth of submerged macrophytes which may then control further phytoplankton development.

Finally, a word must be added on the use of herbicides to control unwanted wetland vegetation. Weed science can provide the wetland manager with a wide array of general and specific herbicides to cover most control needs. Some of these herbicides are so far known to be free of any risk to other organisms of the wetland biota. This argument and the fact that chemical plant control is often the least expensive control method render herbicide use an attractive option. On the other hand, there are two major counter arguments. Firstly, while there is growing worldwide questioning of the real long-term benefits of the extensive use of chemicals in agroecosystems, it would seem unwise to use such chemicals in natural ecosystems. Secondly, there is truly no risk free chemical and there is no easy way to test the long-term effects on the immense number of species hosted in a natural ecosystem.

Re-establishment of plant communities in drained wetlands

In cases where a wetland has remained dry for a short period of years the mere re-instatement of the water regime may re-establish plant communities since many seeds and other plant propagules will have survived. This is especially true in wetlands which showed a non-permanently flooded type of hydroperiod.

In case where a wetland, especially one with a permanently flooded hydroperiod, has remained dry for a long period, successful re-establishment may require human intervention. Examples: diverting water from vegetated ditches, sowing seeds, planting seedlings and cuttings. It is often necessary to create a nursery in Botanical Gardens to propagate plant material. Botanical Gardens of various missions, sizes and levels of operation are found in several Mediterranean countries. The 1999 Membership Directory of the Botanical Gardens Conservation International mentions 49 Gardens distributed in 12 Mediterranean countries as follows: Egypt 1, France 11, Greece 1, Israel 2, Italy 18, Malta 1, Monaco 1, Morocco 1, Portugal 3, Spain 7, Tunisia 1 and Turkey 2. The total number is greater because some Gardens are not members of this society.

In planning revegetation projects with wetland species one can find useful information in several publications (e.g. Klotzli 1987, Hammer 1992, Wheeler et al. 1995, Kenneth et al. 1998, Westlake et al. 1998).

Evaluation of success

The evaluation methods usually applied for both recovered and re-established vegetation are:

- belt transects, especially recommended for forested wetlands,
- replicate quadrats for herbaceous vegetation,
- multiple quadrats for shrub vegetation (Erwin 1997).

Mapping of vegetation pattern, rarity, zonation, biomass, and productivity should be carried out next.

Stabilization

This is the most difficult part of restoration projects. It is necessary that plants should grow vigorously for at least one season while also maintaining minimal nutrient loadings, and the new system should be appropriately monitored. Special attention should be given to controlling the exotic species. In restoration projects where biomanipulation is applied, fish must no be re-introduced until the submerged plants are fully established.

Two cases of vegetation restoration from Israel

Case 1: Agmon wetland-Lake Hula

The drainage of Lake Hula and of the surrounding swamps, in Northern Israel, during the late 1950's resulted in the loss of a very diverse and rare ecosystem. The authorities decided to reflood one part of this wetland in 1994.

The distribution of macrophyte taxa in the wetland has been qualitatively studied since 1995 and their biomass measured since 1997 (Kaplan 1997-2001). The dominant species are: *Ceratophyllum demersum,, Potamogeton nodosus, P. berchtoldii, P. pectinatus, Najas marina, N. minor Typha domingensis, Phragmites australis* and *Chara braunii* (Kaplan 1997-2001, Kaplan et al.1998).

A seasonal pattern of onset (March-July) and offset (August-November) of aquatic (totally and partly submerged) plant growth was observed. An onset of *Phragmites australis* and *Typha domingensis* occurred from late 1994 to mid 1996, followed by a subsequent decline. The peak of submerged macrophyte biomass occurs in June-July and in November-December. All plants (except *P. australis* and *T. domingensis* where existed) are degraded and have disappeared. All macrophytes, excepting stands of *Typha* and *Phragmites*, grew below the water surface under a canopy of floating leaves of *Potamogeton* sp. The decomposition of plant material starts in May-June when the benthic algae die off. Degradation continues intensively into the summer and autumn.

Within the first 2 years of reflooding, 74 plant species colonized the wetland spontaneously. Five out of 11 species designated for re-introduction were successfully established. *Cyperus papyrus* was re-introduced from seedlings and rapidly became the dominant species. *Cynodon dactylon* has established spontaneously. Re-introduced *Nymphaea alba* plants were established only in enclosures protected from grazing. The results demonstrate a high potential for the successful re-establishment of most of the original Hula macrophytes either through spontaneous colonization or introducing of locally extinct species.

Case 2: Yarqon river

The Yarqon river is the southernmost perennial river in the coastal plain of Israel. This river meanders along 28 km through the Tel Aviv metropolitan area and flows into the Mediterranean. Until the early 1950's, the river was clean. Aquatic macrophytes were abundant, and the vegetation on the river banks was lush. Dumping of wastes into the river and exploitation of most of the river water by the National Water System caused detrimental changes to the ecological balance of the river. As a result, many aquatic macrophytes disappeared, while the presence of others was drastically reduced. From the entrance point of one of the tributaries the Qana river until its estuary to the sea, the Yarqon became a "dead" river. Studies of the impact of pollution on the vegetation of the Yarqon and Alexander rivers showed a definite link between water quality and the abundance of various macrophytes, with a greater plant diversity existing in the clean sections (Agami et al. 1976, Litav and Agami 1976). The activation of a new water purification plant in June 1996 greatly improved the water quality of the upstream sections of the Yarqon river. Our assumption was that improvement in the water quality of the Yarqon will enable recovery of its natural flora and vegetation.

The study on Yarqon river is an attempt to re-examine the relationships between water quality and diversity, growth and distribution of plants, and to study the recovery capacity of plants at those sections of the river where water quality has improved. The influence of various pollutants on plant survival, the sensitivity of various plant species to various levels of water quality, and the recovery rate of aquatic vegetation were examined. Chemical and physical analyses of water quality were carried out at four stations along the river, representing various levels of pollution. The chemical and physical analyses included: Biochemical oxygen demand (BOD), dissolved oxygen (DO), turbidity, temperature, pH, conductivity, concentrations of chloride, sodium, potassium, ammonium, detergents and sulfide. During the day, DO levels at the polluted sections were significantly higher than at the clean sections ($p<0.01$). During the night, the decrease of DO at the polluted areas was drastic. At the polluted sections there were clear gradients of the organic load with the descent down the river. The main improvement of the water quality since the operation of the water purification system consisted in the reduction of the BOD and the turbidity levels and a rise in the DO concentration.

It was found that there is a clear link between water quality and the diversity, abundance and growth of aquatic plants. The establishment and growth of seven species of plants transplanted to four stations along the river were better at the clean sections than at the polluted ones. Species diversity (H), species richness (S) and species abundance were significantly higher in the clean sections than in the polluted ones. Also, the composition of species in the clean and polluted sections was different: among 58 species that were found in the river, only 34 species were found in the clean sections. On the other hand, there were not any species that appeared in the polluted sections that did not appear in the clean sections. A negative correlation was found between the BOD levels, ammonium and water conductivity in the water and the growth of the transplanted plants.

The improvement in the water quality enabled the emergent species to establish themselves in the polluted Yarqon river. The *Cyperus papyrus* and *Scirpus lacustris* plants that were planted in the polluted sections became well established. The establishment of plants which were planted in the polluted water was better than their establishment in the same sections in 1973, when the levels of several pollutants were higher. The floating leaved species did not survive in the polluted sections. Still, a local improvement in water quality in station 2, as a result of nearby small springs, allowed plants of *Nymphaea nauchali* and *Nuphar lutea* to establish and continue growth, though only for a period of several months. The establishment of the emergent species near the clean water springs was impressive and their growth was even better than the growth of the plants in the clean sections of the river. This result points out that plants have a high recovery potential in the polluted sections. One can assume that as long as a moderate level of pollution is maintained, emergent species will be able to establish again on the river banks. The high availability of nutrients in the polluted water may improve their growth as well.

The influence of a single factor contaminator such as detergents, sulfides, ammonium and high organic load was examined on three aquatic plants under laboratory conditions. The detergents and sulfide treatments were the most harmful to the plants. However, detergents and sulfide at their average concentration in the Yarqon did not damage the emergent species but inhibited the floating leaved and the submerged species. The average concentration of ammonium in the river did not cause damage to any of the species. High organic load, which caused low DO concentrations and a high turbidity in the water, damaged only the immersed species.

According to results of phytosociological tables of the Yarqon, 58 species were classified into three main groups, according to their abundance, and their frequency in clean and polluted sections. Ten species were classified as indifferent to pollution or even able to benefit from it, 14 species were classified as being able to tolerate a moderate organic pollution, and finally, 34 species were classified as sensitive to pollution.

Generally speaking, the plant distribution that was found resembled the distribution of previous phytosociological tables of the Yarqon. The phytosociological table along the river indicates a fast recovery rate of the natural flora of the clean section, with the improvement in the water quality and habitat conditions from the beginning of the 90's. The richness and abundance of species in the clean sections were higher compared to the richness and abundance in 1973 and 1994. This improvement enabled the reappearance of 14 species that had grown in the Yarqon river in the past and disappeared about 50 years ago. In a period of 3 years (from 1994 to 1997), 11 new species were added to the river. Almost all the added species grew exclusively in the clean sections. Operating the new water purification system did not affect the recovery of the natural flora in the polluted sections.

It can be concluded that a period of about 3 years is not enough to obtain re-establishment of natural flora in sections of the river that suffered many years from severe pollution. Recovery of plants in the Yarqon does not depend only on the improvement of the water quality but also on water quantity. Its seems that operating a water purification system has a limited influence without the flow of a steady and satisfying amount of water and the cleaning of the Sed from organic sediment. Recovery of plants has a positive effect on various components in the damaged Yarqon ecological system. Aquatic plants have an influence on water purification according to their ability to take up nutrients and heavy metals. Recovery of plants on both banks will provide a natural barrier, which will filter the influences of pollution from a diffusion source. It has a mutual ecological effect: water purification in the river will cause recovery of vegetation, which in turn will contribute to the purification of water. This research showed that the recovery of aquatic plants in the Yarqon is today a realistic possibility. The current improvement in the water quality of the Yarqon enables the recovery of species that were present in the river in the past.

Conclusions

Restoration of wetland vegetation is a relatively new and developing field. Although numerous techniques have been developed in the last few years in order to manage aquatic vegetation, restore it to a certain degree and also to face the consequences of degradation from specific stresses, many of those techniques require further study to improve their effectiveness and to identify situations in which they are best applied. For certain kinds of problems, suitable restoration techniques are lacking.

Key considerations in setting targets specifically for the restoration of plant communities should include a thorough understanding of the abiotic factors and of the ecological requirements of the plant material, a prioritization of the functions which plants should perform and an appreciation of the positive and negative interactions between the wetland and its watershed.

More information is needed on the flora and vegetation of Mediterranean wetlands, especially wetlands hosted in the Southern Mediterranean countries. The use of information from North America and Central and North Europe will be necessary for many years yet but effective use will have to remain a matter of both art and science.

Improved techniques for littoral zone and aquatic plants management need to be developed. Research should go beyond the removal of nuisance macrophytes to address the restoration of native species that are essential for waterfowl and fish populations. Palaeolimnological approaches should be used to identify the past trophic history and to infer whether a wetland has been restored to its pre-disturbance condition.

There is a great need for cost effective, reliable indicators of ecosystem function, including those that will reflect long-term change and response to stress. Ecological and genetic research on rare or endemic vascular plants is urgently needed.

No matter how successful a restoration project, its long-term viability will have to meet, among others, two requirements. Firstly, ensuring funds for monitoring through which corrective actions will be identified and implemented. Secondly, identifying and adopting sustainable management practices both within the wetland and in its watershed. It must be stressed that much more research is needed to identify these practices. Ideally, pertinent research efforts must be coordinated and carried out jointly in several countries of the Mediterranean with a more generous financial contribution from the economically advanced countries of the European Union.

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RESTORATION OF WETLAND FAUNA

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Introduction

Recognition of the importance of fauna initially helped raise interest for wetland conservation, and later for the rehabilitation or restoration of some lost wetlands. Although many fauna-oriented restoration projects have been conducted, this is not a dominant theme in the wetland restoration literature (Young 2000), since the various actions included in the topic (habitat manipulation, reintroductions etc.) are usually not termed "restorations" in the literature. However, it is obvious that any wetland restoration project, among other positive benefits, induces some faunal restoration, if only for "low-profile" (in conservation terms) species: invertebrates, common species of fish and amphibians etc. Taking this into account, "Fauna restoration" will be used later in this chapter to mean the restoration of some specific "high-profile" (in conservation terms) species or taxonomic groups. Restoration will be considered in the broader sense of re-establishing former faunal communities or species abundance, regardless of whether any species has actually disappeared or simply reduced.

Aims and objectives for restoration

For over 2,000 years, wetlands have been lost and degraded throughout the Mediterranean (Finlayson et al. 1992), resulting in many animal species becoming rare, being extirpated from the region or even going extinct (e.g. fish: Crivelli 1996, birds: BirdLife International 2000). In recent decades, a few initiatives have been taken, with varying objectives, to restore animal populations or communities in Mediterranean wetlands. Following minor adaptations, the aims and objectives for reintroductions proposed by IUCN (1995) can be applied adequately to broader fauna restorations:

1. Aims
The principal aim of a wetland faunal restoration should be to re-establish viable, free-ranging populations in the wild, resembling as closely as possible the original ones and self-sustaining in the long term. Particular emphasis should be placed on species or sub-species that have become locally extirpated, or have declined to a great extent. They should be re-established within their former natural habitat and range, and should require minimal long-term management.

2. Objectives
The objectives of a restoration may include, singly or in combination: to enhance the long-term survival of a species; to re-establish a keystone species (in the ecological or cultural sense) or an animal community in an ecosystem; to maintain and/or restore the diversity of animal life in a wetland; to provide long-term economic benefits to the local and/or national economy; to promote conservation awareness.

Most restoration projects have many of the above objectives (MINUARTIA 1993, Roman and Mayol 1997); and faunal restoration itself may be only one of the many objectives of broader wetland restoration (Grillas et al. 2000). It must be stressed that not all faunal restorations aim primarily at faunal conservation: removing alien species harmful to human activities, reinforcing harvested populations to increase yields, or restoring species as tools to promote wider habitat conservation (as in the case of umbrella species) are all valid objectives for faunal restoration.

The limits of restoration

Restoring literally implies to revert back to some “original” fauna. However, a precise description of the initial situation is often lacking, so that the former occurrence of species to be restored may be questionable. For instance, the white-headed duck (*Oxyura leucocephala*) was reintroduced into the Balearic Islands based on a single illustration found in a 18th century manuscript (Mayol 1992). Similarly, the proposed (re)introduction of the Adriatic sturgeon (*Acipenser naccarii*) into the Guadalquivir estuary is currently the subject of a hot debate in Spain, because of the uncertainty on its former occurrence in the Iberian Peninsula (Ruiz et al. 1998, Anonymous 1999, Delgado 1999, Domezain 1999). The deliberate release (or lack of prevention for escape) of various bird species in the Camargue (e.g. the glossy ibis *Plegadis falcinellus*, spoonbill *Platalea leucorodia*, marbled teal *Marmaronetta angustirostris*), where the occurrence of resident populations in the past is not proven definitively, is raising a similar controversy (R. Lamouroux, personal communication).

It is proposed therefore, in agreement with others (Kusler and Kentula 1990, Middleton 1999), that a restoration operation should aim at re-establishing populations that are self-sustaining and self-regulating within the current landscape, *"rather than to reestablish an aboriginal condition that can be impossible to define and/or restore within the current context of land-use or global climatic change"* (Middleton 1999). One practical implication would then be that we do not need to be absolutely sure of the former occurrence of a species in a particular wetland before restoration, so long as it falls within its former range and the conditions are otherwise favorable for establishment. Biomanipulations (e.g. introduction of fish to enhance water quality; see below) are sometimes justified on this basis, as they may involve species never reported in the wetland under consideration, although they were possibly native regionally.

Some basic principles

Any sound faunal restoration project must start with an analysis of the causes for species disappearance or degradation, in order to identify the key issues to be addressed. We propose the following table as a guideline for choosing among the possible restoration tools available to the manager (Table 1). For the most common causes of fauna disappearance or degradation, the restoration techniques that have usually been applied in the Mediterranean basin are listed. Case studies will be presented later. Depending on individual cases, "wetland fauna" may signify an individual species, or groups of similar species, or whole animal communities.

Table 1: Proposed classification of faunal restoration actions according to pre-restoration situation.

For each possible reason why restoration can be considered, we list in italics some restoration tools that have been used, or at least considered in the Mediterranean.

1. The whole ecosystem, i.e. the wetland, has disappeared. ➨ *Restoration of the whole wetland ecosystem; the fauna is one of the elements which will partly restore themselves spontaneously. Faunal elements which do not come back following site restoration to be treated as in Case 2.*
1α. The wetland is still present, but there is a limiting, anthropogenic factor acting on the whole habitat and species present. ➨ *Removal of the limiting factor.*
2. Pollution. ➨ *Restore the quality of the ecosystem first (e.g. water quality).* ➨ *Biomanipulation (in the specific case of eutrophication).*

Table 1: Proposed classification of faunal restoration actions according to pre-restoration situation (continued).

3. Species over-exploitation and/or disturbance. ➨ *Sustainable use of harvested species (regulations).* ➨ *Strict protection (local/national level).* ➨ *Limiting access.*
4. Competition/predation by exotic species, or invasive native species that have undergone population explosion for anthropogenic reasons. ➨ *Eradication or control of alien/harmful species.*
5. Critical habitat for the fauna is lacking or partly inadequate (e.g. breeding/spawning sites, water level, feeding grounds). ➨ *Specific habitat manipulation/management.*
6. In general: Legislation insufficient/inappropriate. ➨ *Adapt legislation.* Weak support from local stakeholders. ➨ *Public awareness actions.* Lack of knowledge on targeted fauna and habitat requirements. ➨ *Management-oriented research before action.*
7. There is a limiting factor inherent to the species/group targeted or to surrounding areas (e.g. isolation from potential source populations), whether natural or anthropogenic, which makes natural expansion/recolonization unlikely. ➨ *Reintroductions, translocations, supplementations.*

Some of these restoration approaches have been given a high profile, especially reintroductions (IUCN 1995) and the eradication of alien and invasive species (e.g. Orueta and Ramos 1998 for a major review for Europe).

Examples of fauna restorations from the Mediterranean

Examples are classified and numbered according to the guidelines proposed above. For each situation that may require faunal restoration, the type of action(s) is described and one or two case studies are summarized. A brief review of other Mediterranean examples in various faunal groups is provided. As faunal restoration is still more in the hands of managers than scientists (Allen et al. 1997), most case studies were not published in scientific journals, but instead found in unpublished reports, popular literature etc.

1.The whole ecosystem, i.e. the wetland, has disappeared

Restoration of the whole wetland ecosystem is of course the vital pre-requisite. A case study has been developed elsewhere in this volume (see Chapter XXX) for the Vistre Marshes in Southern France, which were recently recreated 30 years after initial drainage. Although not carried out only for faunal restoration, this restoration effort allowed a large colony of tree-nesting herons to recolonize the area. In just 4 years it has become one of the largest in France (Grillas et al. 2000). This recolonization was possible both because the area is immediately adjacent to the Rhône Delta, one of the key areas in the Mediterranean for these species (Hafner 2000), and because the project also enabled the local re-establishment of those fish populations serving as the main food for these herons (Grillas et al. 2000).

A similar fish population restoration was recently carried out in the formerly drained Hula marsh in Israel, both to attract waterbirds which feed on them (and ultimately ecotourists) and to control mosquitoes (Degani et al. 1998). Proposals to restore whole wetlands specifically for waterbird communities have also been made in Greece (Handrinos 1992).

2. The wetland is still present, but pollution is a limiting factor

Reducing pollution is a prerequisite for restoration of both the wetland and the target species, and management measures are required at the catchment level: better treatment of industrial and urban waste water, reduction of fertilizer use in agriculture etc. Water quality restoration is sometimes carried out specifically for, or at least strongly driven by, single species restoration. The charismatic “flagship-species” therefore brings benefits to the whole ecosystem. Numerous examples are found in Spain with the otter (Prenda and Lopez 1999), and endemics such as the fish *Valencia hispanica* (Planelles 1992, Planelles y Risueño 1995) or the mollusc *Unio aileroni* (MINUARTIA 1993, Carretero et al. 2000). In Slovenia, the Marbled trout *Salmo trutta marmoratus* is one of the most endangered fish species of the Adriatic basin, and pure populations (i.e. not affected by hybridization with the introduced Brown trout *Salmo trutta fario*) are now rare. A management plan for its major remaining population in the upper Soca river was recently designed (Povz et al. 1996). Besides addressing genetic issues, restoration of the native fish population will require maintenance of high water quality in the upper Soca river and regulation of damaging activities that affect watercourses: gravel extraction, logging, dams, water extraction etc. Similar actions to reduce pollution in Lake Mikri Prespa (Greece) and its main tributaries are also planned as part of the restoration plan for the endemic Prespa barbel *Barbus prespensis* (Catsadorakis et al. 1996).

Biomanipulation using fish is one option for controlling biotic responses to eutrophication (Romo et al. 1996). It consists in the control of small, mainly Cyprinid fish which feed on zooplankton, especially *Daphnia* sp., either through direct removal or though releasing piscivorous fish which prey on them. Increased populations of *Daphnia* sp. control phytoplankton, which thrives in eutrophied waters. Algal biomass removal in turn can lead in some situations to clearer water and better growth of submerged macrophytes (Meijer et al. 1999). Whereas biomanipulation has been applied to a number of lakes in Northern Europe, e.g. The Netherlands (Romo et al. 1996, Meijer et al. 1999), its applicability to the Mediterranean basin apparently remains to be tested.

3. Species affected by over-exploitation or excessive disturbance

Over-exploitation of a renewable animal resource, or the repeated disturbance of fauna, can maintain it at levels far below the wetland carrying capacity (Tamisier and Dehorter 1999). Implementation of sustainable harvesting practices, total cessation of harvesting through local and/or national regulations and reduction of environmental disturbance can lead therefore to a spectacular restoration of fauna.

The Etourneau and Ligagneau estates in the Camargue (each ca. 450 ha in extent) used to harbor, respectively, 464 and 207 wintering ducks on average; duck hunting was a major activity. Following land acquisition by the Conservatoire du Littoral, a state Coastal Conservancy body, (in 1983 and 1989 respectively), hunting was banned, access was restricted (but not forbidden) while wardening and wetland management were initiated. In less than 10 years the number of wintering ducks increased 11-fold (to 5,200) and 15-fold (to 3,000) in the Etourneau and Ligagneau estates respectively (Lucchesi and Gerbeaux 1994, Tamisier and Dehorter 1999). In this spectacular restoration of duck abundance, the respective role of the hunting ban, which reduced both direct catch and disturbance, and the new, conservation-oriented wetland management cannot be clearly separated.

Similar cases of bird abundance being restored through legal protection and hunting bans are found elsewhere in the Mediterranean basin, e.g. in the Aiguamolls de l'Emporda in Catalunya, Spain (Sargatal and Fèlix 1989) and in El Hondo, Southeastern Spain, where over 4,000 white-headed ducks *Oxyura leucocephala* were recorded in the summer of 2000, largely because hunting was banned in most of this marsh in the prior 4 years (A. J. Green personal communication). Similarly, fishing regulations (monitoring catches, legislation, e.g. on fishing nets mesh size, and its enforcement) have been proposed to aid the restoration of the endemic Prespa barbel *Barbus prespensis* in Northeastern Greece (Catsadorakis et al. 1996).

4. Competition or predation by exotic or native species

Exotic species always affect wetland fauna, if only through making the species composition of the food web more remote from the "native" one. Fauna may also be affected through competition, predation, or new diseases brought by aliens. Eradication, or at least species control to acceptable levels, is the way to restore the wetland fauna.

Although introduced and invasive species are increasingly being considered as the single most important challenge for nature conservation (Clout 1995, Vitousek et al. 1997), the real impact of most introductions is often hard to quantify. For instance, only 3-4 of the 70 introduced freshwater fish to the Northern Mediterranean region have a proven, or strongly suspected, impact at the species or ecosystem level (Crivelli 1995). Even if the decline of native species is proven, it is often impossible to tell apart the respective role of exotics versus other adverse factors.

Prevention of introductions is of course the best option, and legal provisions are needed (e.g. Galland 1997 for France). Aquaculture operations for the bullfrog *Rana catesbeiana* have been closed by Spanish authorities (Ayllon 1999) based on known impact elsewhere in the world from escapes (Neveu 1997). Similarly, legally preventing the exotic terrapin *Chrysemys scripta* from being imported to Cyprus will help protect the endangered, native *Mauremys caspica* (Hadjichristophorou 2000), by avoiding the presence of a potential competitor.

One of the best-known cases of control of exotics in the Mediterranean is that of the ruddy duck *Oxyura jamaicensis*, a North American species which threatens the white-headed duck *O. leucocephala* through competition and hybridization (Green and Anstey 1992). The latter endangered species is the focus of much conservation effort in Spain (Pereira 1991, Torres and Moreno-Arroyo 2000), and reintroduction plans have been conducted or proposed in several countries (e.g. Corsica: Perennou and Cantera 1993, Balearic Islands: Mayol 1993, Italy: Gallo-Orsi 1998). Control plans for the ruddy duck have been developed and enforced (Anonymous 1994, Hughes et al. 2000), with dozens of birds shot in Spain and France to date and over 1,000 in the United Kingdom in 1999 alone (DETR 2000), out of a population of ca. 4,000 birds.

Other control operations for exotics include the coypu *Myocastor coypu*, which causes damage to crops and water management infrastructures in Southern France (Gindre 2000); over 3,800 have been captured and killed between late 1998 and early 2001 in the 3,000 ha marshes surrounding the Etang de l'Or by a team of 4 commissioned staff (Anonymous 2001). In Mallorca (Balearic Islands, Spain), both the introduced water snake *Natrix maura* and frog *Rana perezi* have been controlled in the habitat of the Mallorcan ferreret *Alytes muletensis*, an endemic toad on which they prey heavily (Roman y Mayol 1997). Local eradication of exotic fish species has been successful worldwide (Knapp and Matthews 1998) and inadvertently happened in Lake Oubeïra (Algeria) for the Chinese grasscarp *Ctenopharyngodon idella* (De Belair 1990). Where total eradication is impossible, management should aim at keeping densities low, but this can have an environmental cost: control of the American crayfish *Procambarus clarkii* in the Ebro Delta (Spain) by farmers with additional, specific pesticides contributes to increased pollution of the delta (Ibañez 1997).

The yellow-legged gull *Larus cachinnans*, although native to the Mediterranean basin, has proliferated due to open-air rubbish dumps. It has adverse effects on other colonial waterbirds, e.g. other gulls and terns, mainly through predation and competition for nest sites (Sadoul 1996). Gull populations have therefore been controlled for several years in France and Spain (Bosch et al. 2000).

5. Vital habitat features for the fauna are lacking/insufficient

Animal species disappear, become rare or fail to reappear following wetland restoration because some key requirement for their life cycle is either missing in their habitat (e.g. breeding and spawning sites, feeding grounds) or inadequate (e.g. water levels). Promoting the natural return of these specific elements (e.g. particular vegetation type or structure), or providing artificial replacements is then the key to the species' return or recovery.

In the Camargue (Southern France), both the Rhône river and the Mediterranean Sea have been contained by dikes since the mid-1800's, and the previous natural process of island creation has been stopped. In the mid-1960's, the greater flamingo, which requires such islands for breeding, stopped nesting in the delta, and it was feared that it had been extirpated permanently through a lack of nest sites. A new island was therefore built in 1970 and decoy nests added in subsequent years (Johnson 1982). Breeding flamingos occupied the site immediately after a 5-year absence from the Camargue and the island has been used every year since. Similar construction of artificial breeding sites for other colonial waterbirds (terns, pelicans, cormorants, herons etc.) in the Mediterranean basin has been reviewed by Perennou et al. (1996), and provide guidelines for both deciding when to make this management decision and practical designs.

Amphibians have also been targeted by such operations, including *Alytes muletensis* in Mallorca, where artificial pools were created in the canyons it inhabits (Roman y Mayol 1997). In Israel, whole amphibian communities are being provided with an ambitious program of artificial pools for breeding, underground road passages and exclusion fences (Ortal 2000). Endangered fish too have been provided with artificial structures, at least in other areas worldwide (e.g. Winemiller and Anderson 1997 in North America), and possibly in the Mediterranean basin.

6. General problems

Any of the previous restoration actions may be hampered by general problems: insufficient or inadequate legislation, lack of local support or lack of knowledge.

The solutions for these, respectively, are to develop and enforce better legislation, to raise the awareness of all stakeholders involved or potentially affected and to conduct additional, management-oriented research.

Inadequate legislation

A legal framework is often a pre-requisite for any management action (McLean 1999). Listing a threatened species in national protection lists may legally bind a state or its provinces to launch specific restoration plans (Criado 2000), as has been demanded for the snail *Melanopsis penchinati*, an endemic of thermal waters of Aragon, Spain (Pinzolas 1999). Legislation is important also at an international level. Examples include the Bern Convention (1979), Bonn Convention (1979), the Convention on Conservation of Biological Diversity (1992), as well as the Birds (1979) and Habitats (1992) Directives of the European Union. All contain provisions to prevent the introduction of exotic species, and to allow their control. The conjunction of new, appropriate legislation at both national and European levels for the protection of the purple gallinule *Porphyrio porphyrio* was the key to its restoration in Spanish wetlands (Gimenez and Viedma 2000).

Insufficient support

Restoration actions may not be possible without local support. Public awareness campaigns about little-known, non-charismatic species (e.g. amphibians, reptiles, invertebrates) can greatly assist restoration plans by making restrictions more acceptable and creating public pressure to allocate funds or cancel damaging projects. This was the case in Spain, with *Alytes muletensis* in Mallorca (Roman y Mayol 1997) and the mollusc *Unio aileroni* in Catalunya (Carretero et al. 2000). Some restoration actions, such as the control of introduced animals may require major awareness campaigns, as they are particularly difficult for the public to accept, and even sometimes for conservationists (Temple 1990, Perennou 1997) because of the need to kill animals, even when it is the only option.

Lack of knowledge

In some cases, fauna cannot be restored because information on the species' management needs is insufficient (Green 1999, Ehrenfeld 2000, Quiros 2000), in which case additional research is a prerequisite. In other cases, information is missing or too dispersed to aid in establishing restoration priorities. Strategic restoration plans established for whole faunal groups at a Mediterranean basin scale (e.g. Crivelli and Maitland 1995, for freshwater fish) can then provide a broad review of current knowledge with frameworks for action. International conventions and national legislation can subsequently rely on them to set their priorities and local projects or specific plans can be developed, as for the Saramugo fish (*Anaecypris hispanica*) (Anonymous 2000a).

7. Limiting factor inherent to the species/group targeted

Even when the habitat is suitable and the species adequately protected, recolonization or recovery may be impossible or unlikely because of poor dissemination abilities, the absence of nearby healthy populations acting as a source, or a naturally low rate of population increase. Restoration then implies the reintroduction or reinforcement of the population. Some simple releases of animals (fish or game species, e.g. ducks) are sometimes presented as "restoration operations" for stocks depleted for various reasons, including over-exploitation. We will not consider these further, as their main aim is usually to provide short-term benefits for hunters and anglers, rather than to restore viable fish or game populations for the long-term (see § "Aims and objectives" above). Since proper restoration usually involves more lengthy, difficult political processes, mere restocking is often favored as a shortcut by administrative units (McGinnis 1994).

Reintroductions involve the re-establishment of a free-ranging, self-sustaining population of a given species that has been locally extirpated, either from animals captured in the wild or via a captive breeding program (Ebenhard 1995, Philippart 1995, Quiros 2000), while a reinforcement is the addition of individuals to an existing population of conspecifics (IUCN 1995). Noting that many of the earlier reintroductions had failed, the Species Survival Commission of the World Conservation Union approved a set of guidelines in 1995 for reintroductions worldwide (IUCN 1995).

Many reintroductions have been or are being carried out in Mediterranean wetlands, especially in Spain, where 9 out of the 14 reintroduction projects funded through LIFE in the whole European Union have taken place (http://europa.eu.int/comm/life/nature/databas.htm). They involve birds, e.g. the white-headed duck in the Balearics and Corsica (Mayol 1993, Anonymous 1997), the glossy ibis, spoonbill and marbled teal in the Camargue[1] (R. Lamouroux, personal communication.) and the crested coot *Fulica cristata* in Valencia, Spain (Montero 2000); mammals, e.g. otters (Anonymous 2000b); fish, e.g. the Samaruc *Valencia hispanica* in Spain (Quiros 2000) and the Saramugo *Anaecypris hispanica* in Portugal (Anonymous 2000a); amphibians, e.g. *Alytes muletensis* in Mallorca, Spain (Roman y Mayol 1997). Using wild caught birds, three separate reintroductions of the purple gallinule *Porphyrio porphyrio* in Spain have resulted in the restoration of healthy populations in S'Albufera de Mallorca (over 200 pairs), the Albufera de Valencia (over 100 pairs) and Aiguamolls de l'Emporda (over 10 pairs), from where natural recolonization of other wetlands in Southern France and the Ebro Delta is proceeding (Sargatal 1992, Gimenez and Viedma 2000).

Projects involving invertebrates are rarer and include the Nayade, *Unio aleroni*, a freshwater bivalve mollusc endemic to northeastern Spain (MINUARTIA 1993). In 1994-95 it was estimated that only 5,000 individuals remained, all in a single 7 km stretch of a stream, Ser, in Catalunya; the species had disappeared from other nearby rivers. The restoration plan initiated in 1993 included captive breeding, infesting native fish with mollusc larvae (a vital phase of its life cycle), reintroduction to good quality streams in nearby protected areas within its former range, and raising awareness in all administrations whose work may have an impact on the targeted streams. These actions are too recent (1997 and 1998) to assess the success of the

[1] For these 3 operations, the respective roles of deliberate releases and accidental escapes are not known. Only the ibis and spoonbill had bred in the wild by 2000.

restoration, but monitoring indicated that the growth rate of the released Nayades is similar to that of the initial population (Carretero et al. 2000).

Reinforcements have also been carried on a number of species: white storks, *Ciconia ciconia*, in Aiguamolls de l'Emporda (N.E. Spain), where the wild population rose from one pair in 1978 to 6 in 1993 and 33 in 1999 through supplementation (Sargatal 1993, Anonymous 2000b); white-headed ducks (Pereira 1991) and spoonbills (Sanchez et al. 2000) in Southern Spain.

Key questions and problems in current restoration projects

Many restoration projects worldwide often inadequately address some important issues during design or implementation stages (Seddon 1999, Ehrenfeld 2000, Fischer and Lindenmayer 2000).

1. Definition of success to be set before the restoration

Logically, success is achieved when the target species or community is self-sustaining at the level and by the date set at inception. However, this is often lacking. For example, the reintroduction plan for the white-headed duck in Mallorca (Mayol 1992) contains no clear target, which makes it difficult today to evaluate whether it was successful. Of the 54 birds released in 2 batches in 1993-95, breeding occurred in 1996 (2 chicks) (Avena and Munoz 1997), 1998 and 1999 (2 chicks) (Gonzalez et al. 1999), and only 4 birds remained in the wild by November 1999 (http://www.geocities.com/lamalvasia/). A similar project for Corsica (Anonymous 1997) established a long-term goal of 50 breeding pairs (which may be unrealistic: see Perennou and Cantera 1993) without giving a time-frame. Success will be hard to evaluate in both cases; however, in the latter case, a short-term objective is provided (20 individuals from the future local breeding center to be reintroduced by 2001, after a test release in previous years), so short-term success can be evaluated soon.

Despite repeated demand for clear, pre-set targets from the scientific community (Ehrenfeld 2000, Fischer and Lindenmayer 2000), knowledge at the preparation stage often is insufficient to help set realistic, measurable targets and an adequate time-frame. When success is not clearly defined beforehand, political considerations may interfere with the technical evaluation of the project, since any restoration can be claimed a success so long as a few individuals of those released remain in the wild, or some breeding occurred, or local public support was gained. In practice, success is often proclaimed, or objectives revised downwards, too early or under political pressure in order to justify the funds invested, to maintain public interest, to get extended support following delays etc. This reflects the fact that restorations are not simply scientific experiments, but complex conservation actions with strong socioeconomic and political components as well.

2. *Monitoring of success often lacking*

This question is linked to the previous one: relevant indicators for success (number of individuals, breeding success, individual growth rate, % occupancy of the area) should be identified and monitored at relevant intervals in order to help the project manager evaluate the progress and adapt to unforeseen events. For instance, restoration of the ferreret *Alytes muletensis* in Mallorca was accompanied by a monitoring program, which enabled managers:

- to assess population changes in terms of size, range extension, area actually occupied and number of sub-populations. All these indicators increased by 8 to 100% between 1991-1997 (Roman y Mayol 1997);
- to detect sub-populations suffering most from predation, identify where breeding success was highest, whether the artificial pools provided were used or not. This helped adapt management on the ground.

Another example is the greater flamingo restoration to the Camargue, which was followed by a monitoring program extended to the whole Mediterranean basin, thus yielding information of high scientific and conservation value (Rendon and Johnson 1996).

3. *No financial accounting*

As pointed out by Fisher and Lindenmayer (2000), costs for reintroductions are published very rarely, which also seems true for faunal restoration. The database of projects funded by the European Union (http://europa.eu.int/comm/life/nature/databas.htm) does provide the cost of operations funded, which run from 331,000 to 1.5 million Euros for three waterbird reintroduction projects in the Mediterranean, of 3-4 years duration each. No measure of success is yet available for these projects.

Cost details would ensure that best use is made of limited conservation money. For instance, the three successful reintroductions of purple gallinules in Spain were apparently done with minimal investment, i.e. capture in the wild, rapid transport, release on the same day (J. Mayol and J. Sargatal, personal communication), which may help other managers plan inexpensive reintroductions of this species elsewhere.

4. *No publication of results*

As mentioned earlier, publication in an easily accessible form is usually lacking for the Mediterranean basin and most information is found in grey literature or popular magazines. The only exception is the journal *Quercus* from Spain, which often summarizes animal restoration projects, regularly referring to original documents such as national Action Plans and providing useful contacts. However, the outcome of most projects remains unknown, preventing useful lessons to be learned from past experience. Reluctance to report on failures or partial successes probably plays an important role.

5. Single-species vs. ecosystem approach

Any restoration project aimed at a single species may affect others. For instance in Spain, it is suspected (though not proven) that the release of purple gallinules into El Hondo may affect the globally-threatened marbled teal *Marmaronetta angustirostris* through reduction of *Scirpus littoralis* on which the teal depends for food (seeds) (A.J. Green, personal communication).

Similarly, it is suspected that high levels of use by greater flamingos may affect other wetland animal species (invertebrates, birds) negatively in both the Guadalquivir Delta (Montes and Bernuès 1991) and other wetlands such as the Ebro and Rhône Deltas (Gallet 1950, Comín et al. 2000). If this is true, then the restoration of flamingoes in the Western Mediterranean basin, with the associated increased use of marshes, may have a drastic effect on the ecosystem.

Therefore, more emphasis should be placed on the ecosystem with all its animal communities, or the ecological processes themselves, even when dealing with single-species restoration. At the very least, the ecological role of that species must be understood and taken into account. The costs vs. benefits should be analyzed before any management measures are taken to increase its population levels further, especially since its former abundance in relation to existing wetlands is unknown.

Conclusions

Faunal restoration in wetlands is a common and diverse practice in the Mediterranean basin at the end of the 20th century, which is conducted for a broad range of potential objectives. Faunal restoration may be achieved indirectly, as a result of broader ecosystem restoration, or may require specific, sometimes highly technical operations, with the control of exotics becoming an ever more critical issue.

However, lessons can still not be learnt fully from the wealth of experiences conducted so far, mainly because of poor reporting, especially on failures or half-successes (e.g. impact of single-species restoration plans on other valuable species, or on the whole ecosystem), inadequate success indicators, insufficient long-term monitoring of restoration projects, and interferences of political considerations with project design or monitoring. Therefore, much remains to be achieved in terms of:

- formulating how faunal restoration should relate to broader ecosystem management,
- monitoring, which implies setting clear and measurable targets from the beginning,
- publishing the results, including failures and details on costs, so as to help others avoid the same mistakes, plan adequate budgets etc.

A wider acceptance of two salient points may be a prerequisite for the latter, by removing the need to "appear successful or perish":

- faunal restorations are experiments which include both scientific and social components, with unpredictable results even when properly planned within current knowledge,
- failures can teach as much as successes, provided that the operations are adequately monitored and their results, whether positive or not, properly understood and publicized.

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PART 2

SITE SELECTION, DESIGN, AND MONITORING OF RESTORATION

IDENTIFICATION OF POTENTIAL WETLAND RESTORATION PROJECTS

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Introduction

Urban expansion, intensification of agriculture, and industrial development during the last century led to the degradation and loss of many wetlands around the Mediterranean (Pearce and Crivelli 1994). The need to reverse these losses, in addition to the recognition of benefits associated with wetlands restoration, have led to the initiation of several restoration projects. Although there is increasing interest in wetland restoration and opportunities are widespread, efforts to restore wetlands are still sporadic, and there is a lack of general planning at the national level (Zalidis et al. 1999). A strategic action plan needs to be developed in order to coordinate and systematize the restoration of wetland resources (Anonymous 1996). The identification of potential wetland restoration projects is an integral part of such an action plan, which aims to locate, prioritize, and ensure the success of future restoration projects.

The objective of this chapter is to present a general procedure for identifying potential wetland restoration projects that can be used for planning and implementing wetland restoration policy at national or even broader scales.

The three phases of identification

A proposed procedure for identifying potential restoration projects can be divided into three phases. The first aims to identify the spatial need for the restoration of wetland functions and to set environmental constraints for restoration in each case. The second phase is more site specific, and evaluates the sustainability of the potential restoration projects through a synthesis of the environmental constraints derived from phase one and the socioeconomic characteristics and particularities of the watershed. This will not only define objectives and actions, but it will provide priorities, costing, funding and social aspects of implementation. The social identification parameters stress the need to link the survival of wetlands with the human dimension to ensure their sustainable management. Phase three is the final outcome, whereby the evaluation of the previous two phases permits the identification and prioritization of potentially sustainable restoration projects. This final phase stems from the need to make sound decisions on wetland management and lead to successful, cost effective projects with broad public acceptance. The three phases are analyzed below.

Phase I: The spatial analysis of restoration needs

The restoration of wetland functions requires answers to the following questions: which functions should be restored, to what extent, and where. These answers are needed especially for the prioritization of restoration needs. The criteria that will be used for this purpose, depend on the level of the performed analysis. Thus, criteria used to identify national needs are not the same with those used to identify the water districts with the most severe problems. The above mentioned questions will be finally answered after the performance of an analysis at a watershed level.

The three levels of analysis

Level One: *National needs*

A complete inventory of all wetland sites of a nation is first needed. It is suggested that the pertinent procedure developed by MedWet (Costa et al. 1996, Farinha et al. 1996, Hecker et al. 1996, Zalidis et al. 1996) should be followed. The drained wetlands must also be included in the inventory preferably with historical information (when available) on their former structural and functional characteristics, past and present uses and the forces which had pressed for drainage.

By appropriately treating the inventory data one may produce national lists with wetlands in various degrees of degradation. Modifiers may be used e.g. rainfall, priority habitats and wild species (according to Directive 92/43/EEC for the European Union countries of the Mediterranean), endangered species, topography, flyways of migratory birds, degree of demand for drinking and irrigation water, water quality problems etc. The results of such data treatments, especially if mapped on national maps, may provide a first picture of the water districts which have the greatest need for lost and degraded functions and values of wetlands. Various other trends may also be revealed.

***Level Two**: Water district needs*

In some Mediterranean countries of the northern coast, the water district is officially or unofficially the unit of management of water and other resources. According to the Water Framework Directive 2000/60/EC, the water district is defined as the area of land and sea made up of one or more neighboring watersheds together with their associated groundwaters and coastal waters.

A water district may comprise one watershed only or several hydrologically interrelated watersheds. A hydrological interrelationship is primarily documented when two or more watersheds show a degree of sharing of their groundwaters.

The analysis of water districts consisting of several watersheds simply aims at ranking its watersheds according to which wetland functions they mostly need. A prerequisite for this ranking is to collect information on the human activities and the water, soil, and genetic resources in each watershed. For example, the establishment of a wetland for water quality improvement in a watershed with intense agricultural development would be far more critical than would be the case in a neighboring watershed with no apparent nutrient runoff problems.

Another factor which needs to be considered in ranking watersheds is the legal obligations of the country with regard to the conservation of particular ecosystems and species hosted in each watershed.

***Level Three**: Watershed needs and wetland assessments*

Through the analyses performed at the two previous levels a list of watersheds will be produced, showing the most urgent need for wetland functions. More detailed information must be collected on these watersheds, firstly, on abiotic and biotic characteristics, activities, land uses etc. and secondly, on their wetlands.

An analysis of the functions of the watershed wetlands must follow through rapid and comprehensive approaches.

Functional assessments should locate wetland functions with the most severe degradation problems, identify those functions that should be restored at the watershed level, to what extent and where in the watershed, and set the general provisions for restoration.

Availability of natural resources and environmental constraints of the water district and the watershed

The creation or re-establishment of wetland functions within a watershed depends on the availability of natural resources. Environmental constraints such as availability of water, landscape morphology, substrate characteristics, flora and fauna are dominant factors influencing site selection for the establishment of wetland functions. These factors not only determine whether a wetland can be restored, but also affect the cost of a potential project. Large-scale excavations, soil transport, structures for controlling hydrology and erosion, and interventions in the landscape for substrate elevation and grading may raise the cost of the project to perhaps unacceptable levels.

For the restoration of a wetland, there are several ecological constraints derived from climate, characteristics of the watershed, structure and functional level of the wetland. Macroscale factors such as climate, landscape topography and parent geology have broad effects on regions. These factors characterize ecosystems based on similarity of inputs (Odum 1983) and establish the type of management that is possible. Although the scientific base is still very incomplete regarding how different ecosystems work and how different environmental factors interact to control functions (Maltby 1991), restoration of wetlands should be viewed within the context of characteristics of the broad area where they are found.

Watershed characteristics, including topography, hydrology, flora, fauna, and land use have both direct and indirect effects on a wetland. The hydrological budget of the wetland, and the biomass and energy inputs are all dependent on features of the watershed and the availability of natural resources. On the other hand, a wetland has a feedback effect on the watershed. This results from the wetland functional ability to provide services and commodities, e.g. wildlife habitats, feed and water for farm animals, water for irrigation, wastewater treatment, and flood control. Restoration projects should consider interactions between the wetland and the watershed in order to set the primary and secondary objectives of the project and to identify the possible negative and positive effects of restoration.

Phase II: Sustainable restoration planning

After locating wetlands where restoration projects should be implemented, sitespecific constraints should be recorded and evaluated in order to identify potential wetland restoration projects and set priorities for restoration. These should be identified at the watershed level and include ecological, technical, social, and economic parameters.

Ecology of the target area

Ecological constraints of the target area

Site limitations provide opportunities to use previous scientific experience on ecological issues, for efficient and sustainable management. Rates of key ecosystem processes, such as primary productivity and decomposition, are limited by temperature, nutrient and water availability, and the temporal variability of these factors as mediated by climate and topography (Chabot and Mooney 1985, Frank and

Inouye 1994). Human intervention may affect the ecological processes but cannot entirely override the ecological constraints of the site. Restoration projects that cannot be implemented and maintained within the ecological limits of each site will not be sustainable and will be costly when viewed from long-term and broad-scale perspectives. Each wetland has a unique set of biotic and abiotic conditions influencing and regulating ecological processes. In addition, the size, shape, and spatial relationships of habitat fragments on the landscape affect the structure and functions of wetland ecosystems (Dale et al. 1999).

Scientific knowledge and technology

Developing a successful restoration plan requires the support of experienced scientists able to identify the problem and to provide the most appropriate solutions. Specific problems usually require specialized scientists. Although wetland restoration is a rather new topic in the Mediterranean basin, there is a considerable number of experienced and specialized scientists working on this issue. The involvement of appropriate people in each wetland restoration project and scientists with adequate experience are prerequisites for the successful design and implementation of a project. The transfer of knowledge, experience, and available technology should also be considered in order to evaluate if a restoration project should be implemented. In many cases, the proposed projects include technical solutions or interventions to the landscape, which require specialized machinery, materials and construction techniques. Projects requiring technologies that are not available or are expensive have less likelihood of being realized. Thus, available technology and scientific knowledge and the possibility of their transfer are criteria that should be incorporated when identifying potential restoration projects.

The socioeconomic aspects of restoration and public acceptance

Since human populations constantly grow, economic activities can produce profound alteration of wetland resources, and resource restoration and management alternatives will be necessary to meet societal demands (Wyant et al. 1995). A socioeconomic analysis of the area should provide formal expressions of socially and culturally desired ecological characteristics based on local needs and constraints. These will identify the need and objectives for additional or alternative restoration actions. In terms of socioeconomic factors, higher priority should be given to the implementation of restoration projects that enjoy public acceptance, that clearly contribute to sustainable development, and that have some assurance of the availability of the resources needed for realization.

The functions and values of Mediterranean wetlands are affected by local societies. Thus, humans should be considered as an integral part of both the problem and the restoration solution. Any restoration project that does not take into account the human factor is doomed to failure (Crisman 1999). The realization of a restoration project depends on the cooperation among landowners and users, public authorities, and politicians (Ramsar Convention Bureau 1999). Therefore, priority should be given to projects with high public acceptance and where wetland restoration is a public demand. Lake Mavrouda in Greece is such a case, where the local municipality requested restoration of the lake, and the restoration project is ongoing. In such cases, restored wetlands are considered capital assets for the local community and part of its culture. As a result, it is easier for human activities to evolve to make wise use of natural resources and thus maintain the sustainability of the restored wetland. Cowan (1999) underlined the role of local community in wetland restoration and pointed out that the most successful restoration projects are driven by social science and participatory management procedures.

Promotion of sustainable development

The sustainability criterion requires that policy-makers address new trade-offs between short-term income objectives and long-term environmental objectives (Plet 1993). Wetland restoration alters the ecological status of the watershed as a result of the feedback effect of restoration. Because of the functions and values of the restored wetland, several socioeconomic changes occur within the watershed including modification of cropping patterns and land uses, creation of new job opportunities and alternative sources of income, and changes in the population structure. The monetary equivalence of the direct uses of a restored wetland, such as hunting, is relatively easy to determine. However, assessment of the added value of the wetland because of its presence in the landscape and the resulting effects on the socioeconomic status and future of the area remain problematic. Thus, the critical question that has to be answered is how the restoration of a wetland in a certain region will contribute to the sustainable development of the area. The answer to this question is complicated and requires detailed studies, including stakeholder analysis (Benessaiah 1998), cost-benefit analysis (Baram 1980), opportunity cost analysis (Committee on Restoration of Aquatic Ecosystems 1992), risk-assessment (Wyant 1995) and a study of project viability. In order to be sustainable, the development of an area must improve economic efficiency, conserve natural resources and enhance the welfare of the people. The above mentioned studies can be used to assist in identifying potential restoration projects that are not only ecologically possible but also desirable in terms of sustainable socioeconomic development in the area.

Availability of funds

The planning and implementation of restoration plans are usually restricted by the availability of funds. The cost of a restoration project is a primary concern for decision-makers. Most of the funds that are used for wetland restoration purposes in the Mediterranean derive from direct financial outlays by governments. However, the implementation of a restoration project may be facilitated in cases where there is financial support from local authorities, stakeholders of the area (farmers, fishermen etc.), and project sponsors such as private companies, non-governmental organizations, and donors. Regardless of the total cost of the project, the availability of funds is considered a critical factor in accomplishing a successful restoration. Financial resources are required to foster the development of restoration projects, and the availability of funds is among the main criteria for identifying potential restoration projects.

It also has to be mentioned that the availability of funds for restoration projects in Mediterranean countries, which are not member States of the European Union, are limited or even non existent. Nevertheless, the authors of this chapter believe that the European Union's Mediterranean countries, should support restoration projects in the others, because the biodiversity of the Mediterranean basin, is a common asset and heritage.

Phase III: Decisions on potential restoration projects

A final decision on potential restoration projects should consider:

- Spatial needs for the establishment of specific wetland functions.
- The impacts of local decisions within a regional context.
- The preservation, or rehabilitation if needed, of the water and soil resources of the watershed.
- A plan for long-term change and unexpected events.
- Preservation of rare landscape elements, habitats and species.
- Avoidance of, or compensation for, the effects of development on wetland functions.
- The presence of land use and management practices compatible with the natural potential of the wetland.

Potential restoration projects should be functionally oriented and should focus on restoring sustainable wetland functions through manipulation of wetland structural components. During the planning stage of a project, the specific needs of the area in terms of wildlife habitats, landscape diversity, and environmental quality should be considered. Environmental impact studies are always necessary to ensure the implementation of an environmentally sound restoration project.

Conclusions

Without a comprehensive reach of watershed assessment, selected restoration measures often ignore underlying problems at a broader scale and are neither effective, nor cost-effective, nor sustainable. Combining information on watershed physical and socioeconomic characteristics, water quality, habitat, land use and ownership, and regulatory jurisdictions with the preliminary analysis of the nature of previous disturbance, allows for the selection of the best strategies to develop sustainable restoration actions and, along the way, suggest the places appropriate for economic development.

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WETLAND RESTORATION USING MODERN TECHNOLOGY IN AGRICULTURAL LANDSCAPES

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Introduction

Agriculture is the largest land use category in the European Union (approximately 40% of total territory). Over the centuries, European agricultural practices have resulted in the unique landscapes common today with their rich variety of semi-natural habitats and biotic communities. However, since the end of WWII, agriculture has also been viewed as a major source of pollution and a driving force behind the disappearance and degradation of Europe's landscape and wetlands. This is because European agriculture has changed notably in the last four decades to become a complex "high tech" industry that has increased agricultural production, thanks to rapid application of new technology and the encouragement of generous support under the Common Agricultural Policy (CAP), by rates far exceeding those of other industries on the continent. Moreover, because of the abundance of agricultural land in and around European ecosystems of particular biodiversity and wildlife interest, the future of the latter is strongly linked with changes in agricultural practices and intensity and ultimately reforms in agri-environmental policies.

To redress the negative impacts of the CAP, recent major initiatives have promoted conservation of existing wetlands and restoration of areas of high and unique diversity, such as wetlands, through the European Union's agri-environment regulation, now a compulsory element of the new rural development policy of the CAP under Agenda 2000. However, guaranteeing success of agri-environment schemes requires an understanding of the relationships between agri-environmental policies, wetland eco-hydrology, and agricultural/wetland management practices.

This chapter discusses the importance of understanding eco-hydrology when undertaking wetland restoration in farmed landscapes, the changes (e.g. lower organic content in the soil, lowering of groundwater levels) associated with both short and long-term wetland degradation, and requirements of wetland restoration in such landscapes. Finally, the uses of eco-hydrological models as complementary tools for assessing wetland restoration prospects in farmed landscapes are discussed.

Role of hydrology in wetland ecology and management

Hydrology is a fundamental factor in the ecology, management and restoration of wetlands. Wetland ecosystems and the factors that have created them (e.g. climate, topography, and geology) are complex, but the most important attribute common to all wetlands is hydrology (Mitch and Gosselink 1993, Gilman 1994). The hydrology of an individual wetland is determined by the system's water balance, which expresses the movement of water into, out of and through it. Hydrologic outflows include evapotranspiration as well as surface and groundwater outflows, whereas major hydrologic inflows include precipitation, flooding rivers, surface flows, groundwater, and tides in the case of coastal wetlands. These hydrologic pathways transport energy and nutrients to and from wetlands. Water depth, flow patterns, and duration and frequency of flooding, which are the result of all inputs and outputs, influence the biochemistry of wetlands and are the ultimate controllers of wetland biota, whether microbial, floral or faunal. According to Mitsch and Gosselink (1993), *"Hydrology is probably the single most important determinant of the establishment and maintenance of specific types of wetlands and wetland processes"*.

Hydrology modifies and determines the chemical and physical properties of wetland substrates (e.g. nutrient availability, soil and water salinity, soil anaerobiosis, pH and sediment properties), which in turn, control specific responses of ecosystem biota (Hollis 1998). Wetland biota are constrained or enhanced by hydrological conditions, and conversely, especially for plants, influence ecosystem hydrology as the wetland matures and diversifies through a variety of mechanisms including sediment trapping, nutrient retention and organic matter accumulation. Thus, a two-way relationship exists between the stability of wetland vegetation and the hydrological stability of a wetland, irrespective of whether the wetland is natural or modified as a result of human intervention. Small changes in water levels in wetlands, or relative changes in the importance of groundwater or precipitation, can lead to significant changes and losses of wetland plants and animal species and ecosystem productivity (Van Wirdum 1986). Hydrology also acts as a limit or stimulus to species richness depending on the hydroperiod and physical energies. At a minimum, wetland hydrology acts to select water-tolerant vegetation in both freshwater and saltwater conditions. Waterlogged soils and the subsequent changes in oxygen content and other chemical conditions significantly limit the biomass and community types of vegetation that can survive in this environment. Thus, the plant species richness of wetlands that sustain long flooding duration is lower than that of wetlands that are less frequently flooded (Kozlowski 1984, Keddy et al. 1994).

The interactions between wetland structure and chemical, physical and biological processes determine the range of functions a wetland can perform. Common wetland functions include flood flow alteration, nutrient retention/removal, sediment trapping, support of foodwebs (Hughes and Heathwaite 1995, Gerakis and Koutrakis 1996). Wetland functions give rise to several goods and services for man and nature, called values, such as biological diversity, water quality improvement, protection against floods. The role of hydrology in these functions ranges from high to very high since hydrology strongly influences all structural characteristics and processes in wetlands.

Agriculture has been one of the main driving forces causing alterations to wetland structural components (soil, water, and vegetation). Land and irrigation water requirements in agriculture have resulted in the degradation and loss of numerous wetland ecosystems in Europe. The aim of this chapter is to demonstrate the importance of hydrology in wetland restoration and to present a modern technological approach useful in the restoration of wetland ecosystems located in agricultural landscapes.

Hydrology-driven habitat restoration

According to Verhoeven (1992), wetland vegetation is the structural and functional backbone of the wetland ecosystem, and it determines to a large extent all other wetland biota. It provides habitat and energy sources to wetland faunal and microbial communities. Although nutrient availability and vegetation management play important roles, hydrology is the factor governing vegetation composition, species richness and biomass. On the other hand, the ultimate control on nutrient levels is determined by the characteristics of the soil (or other substrate types) of the wetland through their influence on water movement, which affects nutrient retention and transport (Wheeler 1999). Moreover, the flows of groundwater and surface water are the most important ecological factors in wetlands. The relationship between hydrology and vegetation in natural wetlands yields insights into the ecological role of water flow and the assessment of measures for restoring degraded wetlands (Ehrenfeld 2000).

There are numerous non-standardized terms and systems used to describe different types of wetlands. The European classification system is based on ionic supply, volume of surface water and nutrient inflow, vegetation type, pH, and peat-building characteristics (Mitsch and Gosselink 1993). As a result, the number of wetland types is highly variable. However, from a plant ecological point of view, three broad types of wetlands can be identified (Wheeler 1999):

- **Permanent**: wetlands (e.g. mires further subdivided into swamps, bogs and fens) in which the normal, small amplitude of water-level fluctuation supports perennial, stable, wetland vegetation.
- **Seasonal**: wetlands (e.g. ephemeral marshes) in which the seasonal amplitude of water level fluctuation supports ephemeral wetland species that can temporarily colonize exposed, moist, substrata.
- **Fluctuating**: wetlands in which the long-term water level fluctuations are of sufficient magnitude and duration (several years) either to cause changes in wetland vegetation composition or to give rise to periods when perennial wetland plants cannot grow.

In other words, water table status (position and fluctuating versus stable), duration and seasonally of flooding, and soil saturation have a strong influence on the composition and stability of wetland vegetation, its distribution and performance, and the wetland's predisposition to mineral-rich and organic soil substrata formation (Mitsch and Gosselink 1993, Noest 1994, Weiher and Keddy 1995, Wierda et al. 1997).

Wetland flora will almost always respond even to slight alterations in hydrological conditions with substantial changes in species richness, diversity and primary productivity (Mitsch and Gosselink 1993, Wassen et al. 1996). This implies that an abrupt, and usually significant, change in the functional integrity of a wetland will result from a change in its hydrology. Changes in the water regime also affect nutrient availability in the soil and thus the primary productivity of wetlands (Wheeler 1999). Observed responses of wetland plants to hydrological changes depend on the species and the magnitude of change and may include a reduced range, or complete disappearance, of species incapable of adapting to a new hydrological environment, persistence in sub-optimal conditions but with reduced vigor, and continued or even enhanced growth in the new water depth range. Moreover, the new hydrological environment may be colonized by new species (Wheeler 1999).

The present flora of the Mediterranean basin is a reflection of a wide range of natural and anthropogenic factors. The plant community in each wetland ecosystem is an assemblage of plant populations that can be divided into two main categories (Finlayson and Moser 1991, Verhoeven 1992):

- Fully aquatic species (hydrophytes) that complete their lifecycle when their vegetation parts are completely submerged or floating.
- Wetland species (facultative or obligate) that spend only part of their life cycle in an aqueous environment or are only partially submerged.

Hydrophytic communities show a broadly similar species composition and structure throughout Europe and the Mediterranean basin. This is in sharp contrast to semi-terrestrial communities, which show marked, mainly climatic, zonation (e.g. peatland vegetation, mosses, grasses and lichens).

Numerous studies have investigated the relationship between wetland plants and water levels. The most comprehensive assessment of individual plants are the works of Ellenberg (1974, 1979), who identified 12 moisture values based on an intuitive assessment of the water preferences of 2000 plants in West-Central Europe (see Table 1) and the works of Londo (1988), who looked at the requirements of plant species in terms of the influence of the water-table. In the Netherlands, many studies examined the response of individual plant species and communities to changes in water regimes (e.g. Grootjans et al. 1988). Londo (1988) used the phreatophyte classification to determine the dependence of wetland flora at a given site on groundwater quantity and/or quality.

Table 1. Description of Ellenberg's (1979) moisture value or water level (F). Occurrences of plant species on a gradient from dry shallow-soil rocky slopes to marshy ground and from shallow to deep water.

F	Description
1	Indicators of extreme dryness, e.g. bare rocks.
3	Dry site indicators, e.g. dry soils.
5	Moist site indicators, mainly on soils of average dampness.
7	Damp-site indicators, i.e. in moist soils which do not dry out.
9	Wet-site indicators, often in water-saturated badly aerated soils.
10	Indicators of soils frequently inundated but free from surface water for long periods.
11	Water plants with leaves mostly emergent.
12	Submerged plants, permanently or constantly under water.
X	Indifferent behavior i.e. a wide amplitude or different behavior in different parts of Europe.
~	In fluctuating moisture conditions.
=	In soils that are regularly inundated.

Other studies have distinguished general trends in groundwater regimes in wetland soils-wetland plants (Ellenberg 1952, Weiher and Keddy 1995). However, specific plant species or communities can occupy a wide range of soil moisture conditions and groundwater regimes, which may be inconsistent between and even within wetland sites (Wheeler 1999). On the other hand, Wierda et al. (1997) showed that the presence/absence of plant species could be an indicator of groundwater regimes in wetlands. Gowing et al. (1998) also established the relationship between soil water-regime and the distribution of selected wetland plants. In general, these results show that mean, maximum and minimum groundwater level and the possibility of inundation during the growing season appear to be the most discriminating parameters. Others (Grootjans and Klooster 1980, Wheeler 1999) found that the magnitude and duration of water-level fluctuations can provide a characterization of hydrological regimes with regard to vegetation composition for situations in which periods of low water-levels control vegetation composition.

Despite the number of studies to interpret hydrological conditions from vegetation data, there is still a demand for more exact and quantitative data on plant species' responses to groundwater regimes and soil moisture status. This is because both changes in the hydrology of wetland sites can give rise to temporary, transient, assemblages of plant species that are overlooked by long-term studies, and the groundwater regime-wetland plant species relationship is geographically limited (Wierda et al. 1997). In addition, difficulties in relating species' distributions and community composition to water table behavior arise for several reasons (Wheeler 1999):

- Most studies are based on correlative methods that assume some form of stable equilibrium between observed water regime and vegetation.
- Hydrological variables (e.g. amplitude, frequency and duration of water table fluctuations), salient to the distribution of water plants can be difficult to identify and characterize.
- Under certain circumstances (e.g. during low water table levels and dry conditions), soil moisture content may be more relevant to plant growth than the position of the water table.

Restoration of wetlands in agricultural landscapes

Given that the major driving force for wetland loss and degradation in Europe has been agriculture, and that agriculture still represents a threat to the remaining European wetlands, the restoration approaches presented in this section concentrate on strategies that aim to reverse the adverse impacts of land drainage and non-sustainable farming practices.

Land drainage and agricultural intensification have been major driving forces in the degradation and loss of wetlands from farmed landscapes in Europe. The scale of wetland destruction has been so impressive in Europe that Hollis and Jones (1991) suggested that the most common wetland types in Europe are "lost", "degraded", and "threatened". Recently (van Dijk 1991, IUCN 1993, Mountford 1994), the indirect harmful effects of unsustainable agricultural activities on nearby wetlands, such as misuse of pesticides, have also been recognized. Thus, it is important to place wetland hydrology and ecology in an agricultural context for three reasons. First, wetland ecosystems are geographically and functionally linked with agricultural ecosystems. Second, wetlands provide multiple values for agriculture. Third, agricultural land occupies approximately 40% of the European Union area. These three points strengthen the argument that agriculture in Europe, as it is most commonly practiced today, still poses a threat to the potential degradation and extinction of wetlands. On the other hand, wetland ecosystems may receive great benefits from agricultural ecosystems provided that the latter are sustainably managed (Gerakis 1992).

One of the major recent initiatives to redress the negative impacts of the European Union's common agricultural policy (CAP) has been to encourage environmentally friendly farming practices through the European Union's agri-environment regulation (EC/2078/92). This regulation was introduced as part of the 1992 CAP reform and offers farmers financial incentives under various schemes, e.g. the Environmentally Sensitive Schemes (ESAs) in the United Kingdom. The schemes, which encourage conservation of existing wetlands or restoration of areas of special biodiversity concern such as wetlands, require an understanding of the relationships between farming, wildlife and agricultural/wetland management to ensure their effectiveness. The aforementioned arguments again reinforce the importance of enhancing our understanding of wetland hydrology and ecology, that is eco-hydrology, in farmed landscapes (Armstrong et al. 1995). This is particularly relevant to wetland restoration schemes in such landscapes. Restoration strategies can include the re-establishment of higher groundwater levels and the adoption of environmentally friendly farming practices (Larson 1991). These can take effect through reducing the impact of intensive agricultural drainage, introducing vegetation control (e.g. mowing, grazing) and sustainably managing water resources.

Wetland drainage and cultivation have changed many features of drained wetlands significantly, including a reduction in the organic content of soils due to oxidation and soil compaction, reduced seedbank diversity and density, and a lowering of groundwater level (Grootjans et al. 1988, Wienhold and van der Valk 1989, Mitsch and Gosselink 1993, Mountford 1994). Organic matter content in restored wetland soils strongly influences surface soil moisture levels and seed germination and survival (Manchester et al. 1998). The use of artificial fertilizers leads to a reduction in plant species richness. Other agricultural management practices such as a shift from hay to silage production have decreased breeding success for birds in lowland wet grasslands (Jefferson and Grice 1998).

The restoration of wetlands that have been long drained and used for agriculture essentially requires a reversal of the changes associated with damage, some of which can be achieved more readily than others (De Mars et al. 1996, Galatowitsch and van der Valk 1996, Wassen et al. 1996, Pfadenhauer and Klötzli 1996, Straskrabvá and Prach 1998, Jepsen 1999). For example, it is often technically easy to increase the height of the water table in drained wetlands and to decrease fertility of the upper soil layers by reducing or ceasing fertilizer application and modifying vegetation management in sites enriched by fertilization (Pfadenhauer and Klötzli 1996). However, physical changes in soil properties can be irreversible and thus require restoration strategies that are based more on rebuilding soil structure and function (Diggelen et al. 1991, Armstrong and Rose 1998).

Treweek et al. (1998) outlined several fundamental factors that must be taken into account when selecting sites for wetland restoration within a farming context. They also defined separate sets of suitability factors for wetland plants and birds. For wetland plants, the most important factors include a control plan for water levels, availability of water, proximity to good quality wildlife habitat, type and intensity of agricultural management and current and historic plant species composition. In the case of wetland birds, restoration suitability factors include the presence of remnant populations of target bird species, the recent or historic presence of target bird species provided sites have not been irreparably damaged, proximity to permanent bodies of standing water, and the existence of a mosaic of existing land-uses, including arable and extensive areas of grasslands, for example, in close proximity to one another, amongst other factors.

Armstrong and Rose (1998) and Diggelen et al. (1995) noted that wetland restoration in drained agricultural landscapes is not just a simple matter of cessation of drainage or reflooding of the area of interest. Wetland restoration also requires active management of both the hydrology and the land (flora and fauna) within an appropriate socioeconomic and political context. Moreover, the requirements of agriculture's continued usage of water and land have to be taken into account, otherwise wetland restoration projects are likely to fail. In addition, Treweek et al. (1998) underline the importance of setting restoration targets to include regional and local variation. This is particularly relevant to wetland plant species that have been restricted or to local distributions outside their natural range and thus would need deliberate re-introduction and management prescriptions. For example, Rodwell (1992) mapped the national distributions of the plant species in England, which make up the various communities of the National Vegetation Classification. This assists in selecting sites with the greatest restoration potential. Sites having the greatest potential to be restored successfully are those that already have high wildlife interest, or are adjacent to existing "good quality" wildlife habitat, or have remained under extensive farming without drainage (Treweek et al. 1998).

Verhoeven (1992) pointed out that in wetland restoration in agricultural lands, it is very important that the nutrient status of the wetland to be developed is in accordance with the type of wetlands being targeted by the remediation. Factors such as soil redox and base status, the nutrient content of flowing water and soil phosphorous, which determine nutrient-related process rates in wetlands, also have to be taken into consideration during wetland restoration (Tallowin et al. 1998). Verhoeven et al. (1993) note that for the conservation and restoration of plant communities occurring in mesotrophic wetlands, it is necessary to manipulate nutrient availability by adpopting measures that remove nutrients from the system by, for example, modifying the hydrological fluxes. Maintenance of a high water table can reduce nitrogen availability by reducing soil-nitrogen mineralization rates. Unfortunately, depleting nutrient fertilizer residues by reducing input alone to achieve comparability with the nutrient status of species rich wetland habitats is a lengthy and uncertain process (Tallowin et al. 1998). Manchester et al. (1998) found that formerly arable sites that have been under intensive cultivation for prolonged periods tend to have high residual soil fertility and an impoverished seed soil bank and are thus relatively difficult to restore to semi-natural grassland communities.

Restoration is further complicated when the land has been subjected to major modifications in the hydrological regime through, for example, under-drainage. Under such conditions, restoration techniques include reinstatement of an appropriate hydrological regime and cessation of fertilizer use. However, in sites where natural processes are not efficient, target species need reintroduction as seed in order to establish wet-grassland plant communities that have been modeled on the vegetation of adjacent wet meadows (Manchester et al. 1998, Tallowin et al. 1998). Table 2 compares the ease with which a restoration project can be constructed and the probability of its success amongst several types of wetlands (Kusler and Kentula 1990, Lockwood and Pimm 1999).

Table 2. Comparison of the reasons of success and failure of restoration projects for different types of wetlands (Kusler and Kentula 1990).

Type of wetland system	Reasons of success	Probability of success (6=highest)	Difficulties associated with restoration
Estuarine marshes	(1) Easy study of hydrology. (2) Experience and literature available on restoration. (3) Small number of plant species to deal. (4) The general availability of seeds and plant stocks. (5) The ease of establishing many of the characteristic plant species.	6	Narrow tidal ranges, unique local conditions or salinity tolerances make certain types of estuarine wetland difficult to restore.

Table 2. Comparison of the reasons of success and failure of restoration projects for different types of wetlands (continued).

Type of wetland system	Reasons of success	Probability of success (6=highest)	Difficulties associated with restoration
Coastal marshes	Reasons of success are similar to those of estuarine wetlands	5	High wave energies and tidal ranges reduce the probability of success of these types of wetlands.
Freshwater marshes along rivers, streams and lakes	(1) Availability of surface water depths from lake or stream gauging records. (2) Literature and experience available on restoration.	4	Complexity of vegetation types, frequent problems with rare species, difficulty in determining underlying hydrology, modifications in underlying hydrological regimes due to activities such as water abstractions.
Isolated marshes fed by surface water	Should mechanisms be available for managing the water supply, then determination and restoration of hydrology is possible.	3	Limited experience and literature available on restoring such wetlands, determination and restoration of underlying hydrology is very difficult.
Forested wetlands	Water regimes evaluated for adjacent waters can be used in cases when records are unknown for such systems. But such surrogate records are not always sensitive enough.	2	Narrow ranges of tolerance render determination and restoration of wetlands very difficult, limited literature and experience in restoring such systems, long period required for development of mature forests.
Isolated freshwater wetlands (fed by groundwater) (ranging from marshes to forested wetlands)		1	Determination and restoration of the underlying hydrology is very difficult, limited experience and literature.

The use of modern technology in wetland restoration

General

In this section, the aim is not to offer an exhaustive list of eco-hydrological modeling examples. The reader is introduced to the subject of applying eco-hydrological models as complementary tools for assessing wetland restoration prospects and examples are provided of some of the main categories of eco-hydrological models. In addition, the status of eco-hydrological models today is discussed to bring to the reader's attention some of the major weaknesses and strengths of these models.

Eco-hydrological modeling: perspectives and examples

Restoration projects are very expensive in terms of both time and money, particularly if they are unsuccessful (Keddy 1999). Mathematical modeling can help reduce these costs and minimize erroneous restoration predictions by using them as an aid to practical restoration projects. First, numerical models provide a means for testing and enhancing our understanding of the fundamental wetland hydrological and ecological processes. Second, they offer a means for testing different restoration schemes for former or degraded wetlands. Thus, mathematical modeling can play a key role in getting the hydrology and ecology right for any practical wetland restoration project.

Any model used in an eco-hydrological study is an eco-hydrological model (Grootjans et al. 1996, Baird 1999). This broad definition includes the plethora of hydrological models that have been used to understand and predict the distribution, extent and frequency of wetland plants and even fauna in relation to the movement of water into and out of the wetland system. These models can, generally speaking, be classified as either conceptually or physically-based according to the physical processes they simulate, and either lumped or distributed according to the spatial description of watershed processes (Refsgaard 1996). The selection of the most appropriate model depends on the phenomena under investigation and the scale at which they are being studied. Baird (1999) asserts that physically-based mathematical models are the most fundamental research tools in eco-hydrology because they are process-based and thus not only enable us to test the theory underlying eco-hydrological processes, but also enhance our understanding of these processes in wetlands.

There are far more examples of hydrological models in literature than there are coupled eco-hydrological models (Barendregt et al. 1993, Poiani and Johnson 1993, Ertsen et al. 1995, Wierda et al. 1997, Wainwright et al. 1999) which directly look at the way water movements affect plants. Moreover, the most widely used models found in literature fall into the category of empirical ordination models (Goldsmith et al. 1986), which although are simple to apply, tend to be site-specific. The reason for this is that many of these models use indicative values that have been derived either from literature or field experimentation, which render them only useful for a limited range of plant species relative to certain factors such as constant moisture and salinity (Gremmen et al. 1990, Mountford and Chapman 1993, Grootjans et al. 1996). On the other hand, more complex eco-hydrological models can deal with soil-vegetation-atmosphere transfers as well as plant growth and seasonal fluctuation in water levels and water quality, thus providing enhanced explanations of eco-hydrological processes (McGaffie and Henderson-Seller 1997, Wainwright et al. 1999).

One of the most complete physically-based, distributed hydrological models is the SHE family of models (Abbott et al. 1986). Al-Khudhairy et al. (1999) applied the MIKE SHE variant (Refsgaard and Storm 1995) of the SHE model to an under-drained agricultural catchment in Southeastern England, both to determine the impact of land-use changes and alternative water management practices on soil moisture storage and to enhance their understanding of the hydrological functioning of the neighboring undrained Elmley Marshes. They found the MIKE SHE model very useful for examining alternative restoration hypotheses. On the other hand, Kaiser et al. (1997) applied the MIKE SHE model to the former Lake Karla in Greece to explore the reason for observed groundwater depletion and to analyze various restoration options, which entail the re-establishment of specific hydrology driven wetland functions, including flood containment, water storage and groundwater recharge. They found that the existing complex hydraulic structures such as weirs, earth dams, pumping stations, all of which tend to be common in managed wetland sites as well as water management practices, could not be accounted for in the MIKE SHE modeling system. This issue was addressed in the SHYLOC project (Shepherd et al. 1999, Al-Khudhairy et al. 2001), where a coupled hydrological/hydraulic modeling system (MIKE SHE/MIKE 11) was applied to the Former Lake Karla. Permanent coupling of one of the most comprehensive physically-based distributed hydrological models with a physically-based distributed hydraulic model creates a complete wetland modelling tool than can enhance the scientific basis for wetland restoration and management.

Examples of current approaches based on linking hydrological models with GIS-based ecological databases are provided by Armstrong (1993) and Swetnam et al. (1998), who applied simple water balance models to study the interaction between ditch and field center by simulating the in-field water balance defined by water table height in the field and ditch water level. They applied these models to investigate the effects of managed ditch water levels for the restoration of wetland habitats and found that it was much more difficult to maintain the wetland status of heavy clay lowlands than that of wetlands with more conductive peat soils. In the case of the former, it was necessary to rely more on modifications to agricultural management to promote successful wetland restoration. Swetnam et al. (1998) also linked the hydrological budget model to a GIS-based database of plant requirements; the hydrological regime predicted by the simple budget model provided input parameters, such as extent and duration of flooding to the wetland geographical information system (GIS) model, which thereafter predicted the occurrence and distribution of species in relation to flood duration, extent and frequency. The authors used the integrated hydrological/wetland GIS modeling system to identify conflicts and to balance the needs of different Environmentally Sensitive Area (ESAs) schemes. This was to enhance management prescriptions in some ESAs, where both botanical and ornithological objectives are ecological targets and are potentially conflicting. Increasing the water table level to attract wading birds on an English wet grassland can lead to deterioration in the grassland due to a decreased abundance of specific species that suffer from water logging.

Another typical example of the application of a loosely coupled hydrological model and empirical-statistical ecological model is given in Ertsen et al. (1995). The hydrological model (one dimensional unsaturated model) provides seasonal fluctuations in groundwater levels as input for their ecological model. On the other hand, the ecological model is composed of four main ecological categories of variables that are considered important: plant species, soil chemistry, groundwater quality and regime, and land management. Although, the model can analyze the ecological effects of various restoration options such as reduced fertilizer application, decrease in groundwater abstraction and changes in land management (e.g. grazing), it, similarly to the remaining existing empirical-statistical models, has the tendency to offer site-specific causal narratives.

Conclusions

There is no doubt that complete eco-hydrological models can play an indispensable role in assessing the effectiveness of a range of wetland restoration scenarios. However, most of the hydrological and ecological models appearing in literature are crude at best. Examples of complex physically-based distributed hydrological models exist in literature, but very few of these have been applied to model wetland hydrology, and fewer yet assess wetland restoration schemes. Examples of applying physically based distributed hydrological models to analyze the effectiveness of alternative water management and/or land use changes scenarios appear to be successful from a hydrological point of view, but show that for a realistic eco-hydrological approach, a comprehensive physically-based model with a hydrochemistry component is indispensable.

The picture is even bleaker for ecological models. Most of the models used in the ecological modeling examples are empirical and based on equilibrium conditions between the occurrence of plants and specific abiotic factors. These static models can only be useful in a stable environment and are of much less value in a dynamic one. More realistic ecological models should predict the time span of vegetation shift and provide true explanations of the response of plant species to changes in abiotic factors. A truly testable physically-based eco-hydrological model with none of the aforementioned deficiencies is a powerful wetland restoration tool.

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GENERAL GUIDELINES FOR WETLAND RESTORATION

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Introduction

Human degradation of Mediterranean wetlands throughout history has made restoration a current necessity, but the practice of wetland restoration is relatively new. Whereas some conservation activities like wetland protection often require making the best of increasingly bad situations, the goal of restoration raises the pleasing prospect of measurable improvement in landscapes (Keddy 1999).

Restoration ecology is a broad subset of ecology, with its own theories, tenets and knowledge, while ecological restoration is the practice of restoring and managing ecosystems (Brinson and Clewell 1993). The practice requires a clear vision of the final expected outcome of restoration, an understanding of the ecological processes needed to restore and maintain the ecosystem and specific skills and techniques to carry out the work. Throughout the Mediterranean region there have been several restoration efforts, of varied success (Zalidis et al. 1999). The most common causes, which prevented full success, include inappropriate planning, partial implementation of the restoration plan and failure to consider the wetland within the context of an evolving watershed. The development of a common, broadly applicable approach to restoration planning is considered essential for the management of degraded Mediterranean wetlands. The objective of this chapter is to provide guidelines for the development and implementation of restoration plans.

Although the guidelines presented here focus on scientific and technical issues, the importance of community perspectives and values should not be overlooked. Wetland restoration should be an open procedure that involves local community stakeholders.

The presence or absence of public support for a restoration project can be the difference between project success and failure. Cooperation with stakeholders and organizations potentially affected by the project can help build the support necessary to get the project moving and to ensure long-term protection of the restored area (USEPA 2000).

Preparation of a restoration project

As every ecosystem has its own special characteristics, restoration plans have to be tailored to a specific project by wetland scientists in cooperation with the agency or consultancy responsible for the field implementation of the project. Each project should have specific and clear objectives and must include detailed descriptions of the technical procedure, cost calculations and monitoring program needed to assess its success. Prognoses for the evolution of the wetland after restoration should be included in the plan along with a follow-up monitoring program. Furthermore, projection of lag time between remedial actions and ecosystem stabilization is critical in any restoration plan.

Preliminary studies

Historical data concerning the structure and functioning of the wetland must be examined to develop a baseline condition for the ecosystem (Steedman et al. 1996). Ecological investigations provide the basis for the technical design and implementation of a restoration project for most types of degraded systems. The analytical parts of these studies include (Bjork 1994) calculation of the hydrological budget and the description of the present ecological status (e.g. physical and chemical conditions, faunal and floral communities, ecological interrelationships).

Chemical data must be collected in order to estimate input (external loading), bioavailability within the wetland and output of nutrients. Additionally, the areal extent of soil types must be mapped. Sediments (and soils) are the memory of the system, from both biotic and abiotic points of view, and influence the functioning of the wetland. These studies should be carried out in conjunction with retrospective palaeolimnological investigations, where appropriate and possible. If sediment is to be removed from a wetland (dredging), it is advisable that limnologists examine the type and distribution of contaminants in the sediment profile, conditions for release and precipitation of essential elements, the behavior of sediments on drying and reflooding and freezing, and the appropriateness of dredged material for soil conditioning in gardens, agriculture etc.

The standard pre-investigation program should assess plankton, macrophytes, benthic invertebrates, fish, birds and herpetofauna, with a special emphasis placed on rare and threatened species. Data on primary productivity should also be obtained to make both quantitative and qualitative comparisons with the system once restored.

The synthesis of the analytical database for the wetland should delineate present ecological conditions, including the relationships between the wetland ecosystem and its catchment, functional aspects and relationships within the system (foodwebs, production and consumption of organic matter etc.) as well as the future development of the system if no restoration measures were taken. In addition, understanding the role of terrestrial-wetland and open water-wetland ecotones is the key to the development of long-term management plans.

Identification of causes of degradation

Restoration efforts are likely to fail if the sources of degradation persist. Therefore, it is essential to identify the causes of degradation and eliminate or remediate ongoing stresses wherever possible. While degradation can be caused by a single direct impact such as water pumping, most degradation results from the cumulative effect of numerous, indirect impacts, such as changes in land use or modifications of agricultural practices in the surrounding area. In identifying the sources of degradation, it is important to look at upstream and up-slope activities, as well as direct impacts on the immediate project site. Furthermore, in some cases (e.g. dam construction) it may also be necessary to consider downstream modifications to system export characteristics, especially hydrology.

Selecting reference condition

Defining a reference condition for ecosystem structure and functions is particularly difficult. Reference sites may be used as models for restoration projects, and serve as a yardstick for measuring the progress of the project. Techniques available for defining reference conditions include historical analysis (Kondolf and Larson 1995), modeling (Camp et al. 1997), ecological analysis (Sagers and Lyon 1997) and multivariate cluster analysis (Harris 1999). While it is preferable to use historical information for sites that have been altered or destroyed, in most cases historical data are non existent. Thus, it may be necessary to identify an existing, relatively intact, similar site as an analog for the restoration project. It is known however, that each restoration project presents a unique set of circumstances, and no two ecosystems are identical. Therefore, it is important to tailor the project to the given situation and to account for differences between the reference wetland site and the one being restored. On the other hand, use of a pre-existing condition of the wetland as a reference level may not reflect current reality. This is especially true where dramatic environmental and socioeconomic changes have taken place, and use of a pre-existing condition may not reflect desirable or obtainable results, for a sustainable wetland ecosystem.

Assessment of wetland functions

Wetland ecosystems are able to perform a number of functions and an analysis of these can provide information to support restoration. During the last two decades, numerous methods for analyzing, directly or indirectly, wetland functions have been developed. Based on the type of information provided, these can be grouped into five categories: evaluations of values, classifications/categorizations of wetlands, characterizations of wetlands, ratings of wetlands, and assessments of wetlands (Hruby 1999).

Evaluation methods are used to determine wetland values in monetary terms based on different types of economic models (Titre and Henderson 1989) or willingness to pay (Farber and Constanza 1987). Therefore one can only indirectly draw a broad picture on functions from which the wetland functions have been derived.

Classifications/categorizations of wetland types are based on shared characteristics. Although not linked directly to wetland functions, they can provide a basis for the development of assessment methods while they provide, directly or indirectly, information regarding the indicators needed for functional assessment (Brinson 1995). A widely known classification in the United States is the one proposed by Cowardin et al. (1979).

Methods of ***characterization***, such as the Oregon method (McCannel 1993), are used for grouping wetlands into those that provide, are able to provide, and do not provide specific functions. They do not yield information about the level of performance of wetland functions.

Rating methods qualitatively characterize the performance of wetland functions, using grades such as high, medium, and low. The most commonly used rating method is the Wetland Evaluation Technique (Adamus et al. 1987).

Assessment is one of the most commonly misused terms. Using an assessment method, numbers are generated to estimate the performance of a function. Among the most recently developed assessment methods are the Hydrogeomorphic (Brinson 1995) and the Indicator Value (Hruby et al. 1995) assessments. In the literature, the terms “evaluation” and “assessment” are often used interchangeably.

Wetland degradation is reflected through the performance of wetland functions. Functional assessment, using the appropriate physicochemical and biological indicators, is a necessary step for wetland restoration and aims to identify which functions are degraded and which should be restored. The final decision on which functions should be restored is based on the outcome of a synthesis of environmental and socioeconomic factors. These set the priorities that should be followed in a restoration project. In other words, this synthesis indicates which of the degraded functions should be restored and to what degree.

Development of project objectives

An essential prerequisite in formulating a restoration project is to set up restoration objectives for the site. Identification of potential wetland restoration is essential to develop the necessary arguments for conducting the project and predicting future status and use of the wetland. The question, for what purpose do we want the wetland, must be addressed in light of the potentiality and constraints of the area. A watershed has the capacity to become only what local abiotic and biotic conditions will support.

Establishing restoration goals for a waterbody requires knowledge of the historical range of conditions existing prior to degradation and what future conditions might be possible. In some cases, the extent and magnitude of watershed changes may constrain the potential for restoration of the site. Accordingly, restoration planning should take into account any irreversible changes in the watershed that may affect the system being restored and focus on restoring its remaining potential.

However, it must be noted that goals in the development process should take into account the results of the assessment of existing and desired wetland structure and functions and integrate them with important economic, social, and cultural values. Deciding among the numerous choices of restoration goals remains problematic because of conflicts of interest among stakeholders of the watershed. Weighing alternatives and ultimately choosing among trade-offs is fundamental to the process of setting goals for wetland restoration in pluralistic societies (Wyant et al. 1995).

Developing countries face the most daunting challenges. Because of their weak economies based on natural resources, they do not have the luxury to sacrifice present welfare for the benefit of future generations. Policy makers need to address new trade-offs between short-term income objectives and long-term environmental objectives (Von Pelt 1993).

A clear understanding of the objectives and performance standards for wetland restoration projects is a critical step in the restoration success. Objectives direct implementation and provide standards for measuring success. Simple conceptual models are useful starting points to define problems, identify the types of solutions needed, and develop a project strategy and goals. Restoration teams should evaluate different alternatives to select those best able to accomplish project goals. The objectives should be achievable both ecologically, given the natural potential of the area, and socioeconomically, given the available resources and the extent of community support for the project. All parties potentially affected by the restoration should understand each project objective clearly to avoid subsequent misunderstandings. Clear, achievable, and measurable objectives increase project efficiency.

Planning and designing considerations

As mentioned, during the planning stage it is critical to focus on whether the proposed restoration activity is feasible and accounts for scientific, financial, social and other considerations. Restoration should re-establish, insofar as possible, the ecological integrity of degraded ecosystems. An ecosystem with integrity is a resilient and self-sustaining natural system able to accommodate stress and change. Restoration strives for the greatest progress toward ecological integrity achievable within the current limits of the watershed, by using designs that favor the natural processes that have sustained local ecosystems through time. Planners must work to ensure a logical relationship among problem and opportunity statements, restoration objectives, and design.

The selection of technically feasible alternatives and the subsequent design is intended to solve the identified problems. The most efficient approach is to conceptualize, evaluate, and select general solutions or overall strategies before developing specific alternatives. In an alternative design, there are three main factors to consider:

1. *Managing causes vs. treating symptoms*. If the causes of impairment cannot realistically be eliminated, it is critical to identify what options exist to manage either the causes or symptoms of altered conditions and what effect those might have.
2. *Water district/watershed vs. wetland site approach*. Characterizing relationships between wetland and watershed requires a good inventory and analysis of conditions and functions. The restoration design should not only include the best solutions, but also innovative solutions for the reestablishment of wetland functions and prevention of impacts on the wetland from land uses in the watershed.
3. *Spatial and temporal considerations*. Over time, variation among watershed elements provide differential opportunities for desired functions. Dynamic equilibrium requires that the restoration design is afforded an opportunity to mold itself to both changing conditions over time and disturbances that are part of the natural environment.

These factors may determine whether a passive (non-structural) alternative restoration approach is more appropriate than an active one. Once restoration alternatives have been defined, the next step is to evaluate all feasible alternatives and management options. For this reason, some basic supporting analyses are needed for selecting the most appropriate alternative including:

- Feasibility study.
- Cost-effectiveness study.
- Risk assessment.
- Environmental impact analysis.

Wetland restoration plans should be developed in accordance with local and national regulatory frameworks. In all Mediterranean countries, environmental permits are necessary for the implementation of restoration projects. However, in many cases the objectives and success criteria are not clearly stated, and mechanisms are not incorporated in the regulatory permit to ensure compliance with restoration plans and to provide for mid-course corrections if plans fail to achieve their intended results (Committee on Restoration of Aquatic Ecosystems 1992). Knowledge of national and local jurisdiction allows the planner to understand extrinsic forces that must be accounted for in the development and implementation of a project. While not always directly related to the development of management plans, awareness of current environmental legislation can help planners to recommend appropriate courses of action (Emmer 1991).

Restoring the structure and functions of the wetland

Many wetlands in need of restoration have problems that originated from harmful alteration of their characteristics, which in turn may have led to problems such as habitat degradation, changes in flow regimes, and siltation. Stream channelization, ditching in wetlands, disconnection from adjacent ecosystems, and shoreline modifications are examples of structural alterations that may need to be addressed in a restoration project. Both its distinctive structural components and functions define an ecosystem. Restoring the functions of a wetland is usually easier than restoring its structure. In order to maximize the societal and ecological benefits of a restoration project, it is essential to identify what functions should be present and to make the reimplementation of missing or impaired functions a priority in the restoration. In most of the cases, the restoration of wetland functions usually depends on interventions at the wetland, and watershed or water district scales.

Several techniques and methods can be used for the restoration of degraded wetlands. Each restoration plan is site specific, and therefore the applied methods and techniques may vary from site to site according to the objectives of restoration and the particularities of the area. Restoration techniques and methods can be classified broadly as hydrology, chemistry/sedimentology, and biology (Wilcox and Whillans 1999). **Hydrological** restoration usually refers to the establishment of desirable hydroperiods in the wetland ecosystem. Construction of dams, barriers, dikes, pumps, and ditches are most often used to control and modify the hydrological status of wetlands.

Wetland restoration from a **chemical** point of view refers to the improvement of water and sediment quality. Best management practices (wetland buffer zones, erosion control, sustainable agriculture etc.) may be applied at the watershed level to reduce inputs of contaminants into the wetland. Wetland restoration through sediment management includes control of erosion and sediment inputs from uplands and proper management of dams on tributary rivers. Sediments act as a buffering agent against external disturbances. In many cases, wetland sediments became sinks for numerous pollutants such as pesticides, chlorinated organic compounds and PCB's. Restoration measures in such cases can include the deposition of clean sediment over contaminated sediments (USEPA 1994), isolation of sediments from the surrounding environment, immobilization and volatilization of contaminants (McNicoll and Baweja 1995). Phytoremediation (Cunningham and Lee 1995) and bioremediation (Reeders et al. 1993) may also be considered as alternatives in some cases. Removal technologies, via mechanical or hydraulic dredges, are more widely used. However, these methods should be applied only in shallow water bodies suffering from heavy sediment deposition (Bjork 1994).

Restoration methods with a broad **biological** interest can be classified as (Wilcox and Whillans 1999): 1) Enhancement of populations of target-organisms. For this purpose, methods such as seeding, transplanting, and stocking are employed (Hayes et al. 2000). An important point to consider is that the source and genetic origin of any introduced species should be considered before implementation of such activities. 2) Management of non-target organisms mainly for the reduction of competition. The most common methods used include harvesting, exclusion and hydrological, chemical and biological control (Madsen 1997). 3) Habitat enhancement through vegetation and hydrology management. Existing habitats may be physically modified or new habitats created in order to benefit target species by providing food, nesting, and spawning areas.

Using the watershed approach

Restoration requires a design based on characteristics of the entire watershed, not just the degraded wetland. Activities throughout the watershed can have adverse effects on the aquatic resource under restoration. A localized restoration project may not be able to change conditions in the whole watershed, but it can be designed to accommodate watershed effects within a realistic context. New and future urban development may, for example, increase runoff, stream downcutting and bank erosion, and pollutant loading. By considering the watershed context in this case, restoration planners may be able to design a project for the desired benefits of restoration, while neutralizing or remediating the effects of adjacent land uses on point and non-point pollution. For example, in choosing a site for a wetland restoration project, planners should consider how the proposed project may support related efforts in the watershed, such as increasing riparian habitat connectivity, reducing flooding, and/or enhancing downstream water quality. Beyond the watershed, the water district in the broader landscape context also influences restoration through factors such as interactions with terrestrial habitats in adjacent watersheds, or the deposition of airborne pollutants from other regions. In many cases, two or more countries may share the watershed area of a wetland or even the wetland itself. In such cases of transboundary wetlands, there is a need for interregional and international cooperation in order to promote the sustainable use of the natural resources in the area (Utton and Reclaff 1978).

The special management needs of restored wetlands

It has been well established that integrated, sustainable management of wetlands (and their watersheds) is the prerequisite for their conservation. The principles of such management and pertinent guidelines hold true for all wetland types and for all situations and are described and recommended by the Ramsar Bureau and are adopted by many nations.

One can hardly envisage any existing Mediterranean wetland whose management objectives do not need to include restoration of some structural and functional component to some degree. The management principles, therefore, are the same for all existing wetlands regardless of the intensity of restoration needed. However, there are at least two issues which need increased attention: firstly, design and implementation of monitoring, and secondly, adaptability.

The attention paid to monitoring and adaptability should increase as the intensity of the needed restoration increases. Thus, maximum attention is required in the cases where a wetland is restored on the site of a drained one and even more so on a site where no wetland had ever existed. In these cases a young ecosystem is created which means that succession takes place very rapidly for a considerable period of time. This fact, coupled with the great difficulty to accurately predict the path and timing of succession and the need to achieve the objectives of restoration, requires large investments, in terms of funds, equipment and skilled personnel, and ongoing monitoring. It also requires sufficient funds and personnel to promptly apply the so-called adaptable management.

To ensure monitoring and adaptive management, high availability of funds and personnel is not enough. A very efficient administrative body endowed with the appropriate mechanisms to make quick decisions is also necessary for the restored wetland.

Conclusions

Although it is impossible to plan for the future precisely, many foreseeable ecological and societal changes can and should be factored into restoration design. For example, in repairing a stream channel, it is important to take into account potential changes in runoff resulting from projected increases in upstream impervious surface area due to development. In addition to potential impacts from changes in watershed land use, natural changes such as plant community succession can also influence restoration. For instance, long-term, post-project monitoring should entail successional processes, such as forest regrowth in a stream corridor, which should be taken into account when evaluating the outcome of the restoration project.

Perhaps the best way to ensure the long-term viability of a restored area is to minimize the need for continuous maintenance of the site, especially water augmentation, vegetation management, or frequent repairing of damage done by high water events. High maintenance approaches not only add costs to the restoration project, but also make its long-term success dependent upon human and financial resources that may not always be available. In addition to limiting the need for maintenance, designing for self-sustainability also involves favoring ecological integrity, as an ecosystem in good condition is more likely to have the ability to adapt to changes.

For wetlands which are re-created on completely drained sites or for those created on sites where no wetland previously existed, considerable investments are needed on monitoring and adaptive management.

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PART 3

REGIONAL RESTORATION EXPERIENCE

RESTORING MEDITERRANEAN COASTAL LAGOONS AND PONDS

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The structure and functions of Mediterranean Coastal Lagoons and Ponds

This chapter discusses basic structural and functional aspects of coastal lagons and ponds in the Mediterranean basin, the main antropogenic impacts affecting them and efforts aimed at ecosystem restoration. These two wetland types are distinguished by their morphohydrodynamic characteristics, including origin and evolution of the basin and hydrological features. These criteria are closely linked and sensitive to anthropogenic disturbance. Soils, fauna, flora and landscape are resources that are damaged as a consequence of alteration of the first (Viñals 2000).

Lagoons and coastal ponds are terms often used synonymously, but they have a totally different hydrodynamic behavior (Viñals 1999). Although the English term "lagoon" has been imported to the Mediterranean basin and is used indiscriminately for nearly all coastal-plain wetlands it is not narrowly defined. Lagoon hydrology is controlled mainly by tides. Tidal phenomena in the Mediterranean basin are very weak or absent; therefore, it is rarely appropriate to call a Mediterranean wetland a lagoon. The only wetland systems that function in this manner are the coastal wetlands of the upper Adriatic and those of the Gulf of Gabes in Tunisia where tides are clearly evident. Coastal ponds (called "Albufera" in Spain-derived from the Arabic, "Étang" in France and "Stagno" in Italy) have a hydrodynamic behaviour autonomous from the sea unless they have been connected artificially to the sea.

Marine phenomena have played an important role in the origin of both lagoons and coastal ponds. They are responsible for the formation of spits and barriers that closed topographic depressions associated with coastal drift and transversal currents.

One characteristic of these coastal wetlands that distinguishes them from delta wetlands is their relatively longer life. From deep core records it is clear that these wetlands have developed successively different morphodynamic wetland sub-types from at least the upper Pleistocene (Viñals 1996), and the present model dates them from at least the Flandrian period (6000 BP). The current morphology of the greater part of the existing Mediterranean delta takes place in a historic moment, as it happens in the cases of the Ebro river and the Por river (Bondesan 1990, Ibañez et al. 1999).

Lagoons are leaky, that is, they broadly communicate with the open sea by inlets whose hydrodynamics are ruled basically by the tides that introduce seawater daily, independent of the contribution of other sources. In this manner, tides control the hydrological, morphological and environmental features of these wetlands. These lagoons are morphologically the most diversified wetlands and one of the most widespread land-sea ecotonal systems throughout the world. However, they are rare in the Mediterranean basin due to a lack of significant tidal fluctuation.

Non-tidal coastal ponds are the most frequent coastal wetland type in the Mediterranean basin. They differ from lagoons in that their longevity is independent from the sea (although the sea has exerted much influence on their origin). These are enclosed water bodies having a tight barrier. They can also be connected to the sea by inlets, but these are not tidal channels and practically do not introduce seawater into the pond.

Finally, coastal ponds are endorreic or semiendorreic water bodies (if there are channels discharging to the sea), whose sediment barriers have small connections to the sea. Although mostly marine in origin, these ponds are under the control of continental hydrodynamic processes and water (rain, superficial or in many cases, groundwater) that with time can replace seawater. The longevity of coastal ponds is limited because they tend to infill except when there is some renewal factor such as pronounced sea level rise or subsidence. Many of these ponds are presently freshwater, brackish or salt marshes that are palustrine in character. In lagoons, these environments are frequently along inland shores and interior islands.

There are other coastal wetlands in the Mediterranean of very different origin, evolution and hydrodynamics. This is the case of some endorreic depressions (in tectonic basins, pliocene and quaternary depressions, subsidence basins, sinclines, solution basins), and created wetlands that were originally lagoons or coastal ponds (saltpans, ricefields, aquaculture facilities).

This paper focuses on coastal ponds since they are the prototype of the Mediterranean coastal wetland. The hydrological regime of these systems can be as diverse as their principal water sources. They range from fresh to hyper-saline with water supplied from precipitation, river inflows (baseflow and floods), the sea (percolation and constructed channels), groundwater and anthropogenic contributions. These wetlands lose water in many ways: superficial release to the sea, evapotranspiration, discharge to aquifers and artificial drainage systems. The presence of standing water can be permanent, seasonal or episodic depending on the water source, local climate, soil and vegetation. These factors must be considered when restoring a wetland to its natural dynamics.

Degradation of Mediterranean coastal wetlands

Coastal ponds are the one Mediterranean wetland type that has suffered the most degradation through time, especially those systems used as a freshwater supply. Major lagoons around the Mediterranean are located close to densely populated and economically important areas, and at the edge of large agricultural plains, with expanding populations and a potential for intensive/extensive agriculture and tourism.

Activities generating the greatest pressure on wetlands are linked to agricultural use and urbanization (for residents or tourists), where disputes arise over resources, physical space and water.

During the 1950's, the use of the Mediterranean coastal wetlands began to change. In the Camargue, 40% of the natural area was lost between 1942 and 1984; in Italy, less than 100,000 ha of the 700,000 ha of coastal marshes existing at the beginning of the century, remained in 1994. These freshwater wetlands were transformed into ricefields, a crop that is perhaps compatible with the wetland ecology. However, with the rice production crisis of the 1960's, ricefields were changed into other irrigated agroecosystems (e.g. maize, sugar beets, alfalfa), which interact less positively with the neighboring natural wetland ecosystems. Moreover, land reclamation works and other interventions to soil and water resources expanded, resulting in increased inflow of pollutants of agricultural origin into the wetlands, and disturbances in both the hydrogeological balance and sedimentation processes. This resulted finally in a great net loss of wetland area along Spanish (Albufera of Valencia, Marjalería of Castellón, Marjal of Almenara) and Italian coasts (Friuli-Venezia Giulia) (Musi et al. 1992) and elsewhere.

Additionally, shoreline areas were urbanized for industrial, residential and tourist uses due to the low price of wetlands. This led to:

- Serious wetland areal losses (for instance the industrial complex of Fos close to Marseille occupied 4,600 ha of the wetland). Use of the barrier formations for tourist development (these ecological units have all been uniformly transformed throughout the Mediterranean from the Bardawil Lagoon in Egypt to the Spanish coast).
- Water pollution, which has become a serious and growing problem. Coastal ponds and lagoons have been used for dumping wastes, with prevalence of economic over ecological criteria.

- Weakening of the coastal barrier due to increased marine erosion. Reduced availability of sand along the coasts, generally as the result of river damming and/or sand mining from the stream, is particularly noticeable in the great Mediterranean deltas (e.g. Po, and river Nile). Construction of fixed coastal defense structures (seawalls, groins, detached breakwaters, jetties) generally has interfered with nearshore sand drift, inducing beach erosion. This is a very serious problem in the barriers of the upper Adriatic coastal wetlands such as the Lido di Venezia in Italy. Another way of weakening the barriers is opening new inlets for tourist use with a resultant change in the hydrological regime. This is the case of L'Étang de Salses-Leucate en Perpignan, France (Roux and Tesson 1992).
- Infilling of coastal ponds and lagoons due to sediment and organic matter accumulation coming from the catchment. This has happened at La Laguna di Venezia where the pollutant load from a human population of 20 million living in the drainage area (Sestini 1992) far surpasses the self-cleansing capacity of the lagoon by tidal flushing from the Adriatic.

This situation continued until the end of the 1970's, when the protection of natural areas according to ecological criteria became general in Southern Europe. Many wetlands began to be protected based on their biodiversity value (for example, in Spain, Doñana was protected under the regime of National Park in 1978).

Table I presents the principal impacts affecting coastal Mediterranean wetlands, the ecosystem response to these and the common corrective measures to restore ecosystem functioning.

Evidently, many are the actions that can affect one or more of the resources of a wetland. Usually, they also generate synergy effects and anything affecting water, for instance, has immediate repercussions on flora and fauna. Besides, in freshwater wetlands, the most frequent conditions affecting the quality of resources are related to agricultural activities and to industrial waste also. Brackish or sea water wetlands are related to mining and, in the worst cases, to their transformation into building lands.

Regarding the wetland morphology, the changes in the structure of the basin are mainly related to a decrease in its capacity to hold water, due to the reduction of the swampy perimeter. These actions are a result of traditional agricultural uses and, more recently, due to dumping of solid waste.

The physical and chemical properties of soil have been altered and they have gone through a process of compaction and the subsequent subsidence.

In saline coastal wetlands, a complete control of the morphology of those wetlands exploited as salt pans is to be found; otherwise, most of them have been infilled.

Table 1. Main impacts on wetlands.

Affected Resource	Activity	Effects on wetland	Corrective measures
WATER	Aquifer depletion	Decreased water supply. Salinization of coastal aquifer due to salt water intrusion. Induced subsidence	Sustainable exploitation of groundwater
	Artificial overflowing water	Creation of flooding areas Increased salinity and eutrophication	Maintenance (dredging)
	Irrigation residual water and sewage spills	Eutrophication Salinization Microbial pollution	Application of residual water treatment plans. Wiser use of fertilizers
	Industrial and mine wastewater spills	Toxicity (bioaccumulation) Salinization Acidification Destruction of aquatic ecosystems Water rendered useless as a resource	Application of water treatment systems in industry and mines. Construction of residual and industrial water collectors along the perimeter of the wetland
	Solid waste	Toxicity of surface and groundwater due to leachates. Destruction of ecosystems Water rendered useless as a resource	Prohibition of potentially dangerous solid waste disposal in groundwater recharge areas and in the wetlands
	Dam construction upstream	Reduction of sedimentation rate and volume of water supply reaching the wetlands. Erosion and disappearance of some wetlands linked to fluvial morphologies (especially deltas)	Agreement on ecological water flows
	Hydraulic and communication facilities	Reduced groundwater recharge rate. Increased wetland drainage speed. Increased run-off and erosion	Maintenance of traditional channels (natural bottoms and banks). Designing facilities avoiding wetlands

Table 1. Main impacts on wetlands (continued).

Affected Resource	Activity	Effects on wetland	Corrective measures
LANDFORMS	Infilling for farming and urban uses, and to provide facilities and services or to develop leisure activities and tourism	Destruction of morphology, vegetation and habitats. Reduced groundwater recharge rate. Loss of flood control capacity. Indiscriminate increased access to the wetland. Decreased humidity and rainfall. Disappearance of the wetland in extreme cases	Absolute prohibition of practices leading to infills of the wetlands
	Deepening of wetland areas (extraction of peat, sand)	Altered morphology and changes in hydrology and habitats. Occasional increased erosion	Control of extracting activities by EIA Actions leading to restoration and recovery if necessary
	Hydraulic and communication facilities	Fragmentation of territory Infilling of wetland areas	Designing facilities avoiding wetlands
	Infilling with rubble and other solid waste materials	Disappearance or seepages and sheet flow. Increased pollution risk of sediments and soils (due to leachates)	Control and clean-up of dumps. Prohibition of dumping
SOIL	Depletion of aquifers	Salinization Acidification Subsidence	Sustainable use of aquifers
	Redirecting flows	Oxidation and compaction. Changes in physical, chemical and biological characteristics of soil	Prohibition of wetland drainage. Restoration measures
FAUNA	Over-hunting	Reduced waterfowl population. Altered natural distribution of species. Lead poisoning	Application of regulating and wise use plans
	Over fishing or inappropriate fishing techniques	Reduced fish biomass in general and particularly of those of commercial value	Application of regulations and wise use plans

Table 1. Main impacts on wetlands (continued).

Affected Resource	Activity	Effects on wetland	Corrective measures
FAUNA	Aquaculture	Destruction of natural habitats. Damage to native non-commercial species. Alterations in water exchange with the sea (limitation of inlets). Water pollution from feed and fish excreta and sanitary products	Definition of protective perimeter of exploitation. Construction of wastewater treatment plants
	Hydraulic and communication facilities in the wetland	Isolation of ecosystems	Designing facilities avoiding the wetlands
	Introduction of exotic species	Detrimental to native species and loss of biodiversity	Designing facilities avoiding the wetland
VEGETATION	Deforestation of catchment areas (increased erosion)	Increased infill rate in wetland and loss of flood control capacity	Reforestation of catchment areas. Planting riparian vegetation
	Introduction of exotic species	Detrimental to native species and loss of biodiversity	
	Uncontrolled fires (chance or intentional) in wetland	Indiscriminate loss of biomass, habitats and ecosystems. Decreased rainfall, increased erosion	Promotion of natural regeneration. Restoration of vegetation
	Hydraulic and communication facilities	Local loss of plant biomass	Designing facilities avoiding the wetland
LANDSCAPE	Infill for farming	Loss of landscape quality	Replacement of crops not compatible with conservation. Restoration actions
	Urbanization	Loss of landscape quality	In existing urbanization, possible restoration actions
	Solid Waste Disposal	Loss of landscape quality. Increased sanitary risks (rats)	Clean-up of dumps. Control of illegal dumps. Prohibition of dumping

Flora and fauna suffered the consequences of the alteration of the components of the biotope. But also, there are certain impacts that affect them specifically, such as the introduction of exotic species and deforestation or the decrease of populations due to over-hunting or over-fishing.

Finally, the degradation of the aesthetics of the landscape must be mentioned. This can be noticed in many coastal wetlands due to the construction of structures and buildings of dubious aesthetic quality, in no way in harmony with the cultural tradition.

Preventive and corrective measures that should be considered in the rules concerning territorial ordination are proposed. Concerning restoration techniques, the paper from Giró (in press) comprises many uses that could be implemented in Mediterranean coastal lagoons, since some of those uses have already been successfully tested in other environments.

Restoring coastal wetlands

It is essential that restoration programs define the end situation to be reached. Frequently, there is little idea of the pre-impact condition of the system in order to set a goal, and far too often project managers are afraid to state so. If, by chance, one finds a previous condition having the desired stability, he would be facing an important challenge. These ecosystems have undergone ceaseless transformations in the past six thousand years. Human modifications also affected the evolution of ecosystems due to secular interventions carried out in the wetlands to adapt the environment to meet human requirements (economic, political or social) that have evolved through time.

Basic research is needed to delineate past ecosystem conditions. Palaeoenvironmental reconstructions can show the evolution of coastal wetlands via sediment analysis of pollen, subfossil shells and invertebrates, together with absolute dating to establish time sequences. Archaeological remains, historical records and ancient cartography allow us to reconstruct the history of human interventions in the region and gauge the degree of modification that has been inflicted on the natural evolution of the systems.

With respect to implement restoration projects in coastal wetlands, there has been significant experience for several decades in the United States (Lewis et al. 1995). The basic aims that guided these actions have been the protection of the coastline and wastewater treatment. The philosophy underlying how these matters have been treated has been very clear; the *"No wetland loss"* principle has been interpreted freely, creating wetlands to compensate for destruction of natural ones where pressure on the wetland was too strong to resist. This principle has not had the success expected, since it is very difficult or impossible to duplicate the values and functions of a wetland in another place (Mitsch and Gosselink 1993). However, this has allowed advanced development of environmental restoration engineering based on the concept that wetlands must evolve with as little human intervention as possible (natural viability). Nevertheless, on many occasions excessive engineering action has been observed.

In Northern and Central European countries, restoration programs since 1980 have focused on the classic subjects of re-vegetation, restoration of habitats and hydrological restoration. This effort has produced a rich literature during the 1990's, including case studies, scientific training and restoration engineering techniques (Finlayson and Larsson 1990, Eiseltova 1994).

Prior to 1990, all efforts in the Mediterranean basin were focused on conservation. It was considered essential to save and protect what still existed before trying to recover major areas lost during previous decades. This policy was not only the basic guideline followed in the European Mediterranean, it was also the fundamental criterion in the management of African Mediterranean wetlands, where commissioning restoration projects implied social conflict due to increased water demand for conservation uses to the detriment of populations already facing shortages (Dakki and Aziz el Agbani 1995).

The Grado Strategy of 1991 was one of the first documents referring explicitly to a mandate that *"priority sites for wetland restoration be identified and techniques be developed and tested for their complete rehabilitation"*. Although some of the work presented in the XXXVth Executive Board Meeting of the International Waterfowl and Wetlands Research Bureau (Finlayson et al. 1992) concerned restoration projects for Mediterranean coastal wetlands, most continued to be dominated by conservation management issues.

In 1995, a meeting was held in La Rabida (Huelva, Spain) specifically on restoration: *"Ecological basis for restoration of wetlands in the Mediterranean basin"*. This meeting reviewed progress in the field and how American experiences might be extrapolated to the Mediterranean basin feasibly. Interest in this subject has grown to the point that the technical session of the First Meeting of the MedWet/Com1 held in Thessaloniki in 1998, was devoted to restoration.

Several projects have already been undertaken in the Mediterranean during the 1990's, especially in European countries: Le Foci dell' Isonzo (Italy), Burano (Italy), Laguna di Orbetello (Italy), Aiguamolls de L' Empordà (Spain), S' Albufera de Mallorca (Spain), La Albufera de Valencia (Spain), Mar Menor (Spain), Marais de Vistre (Camargue, France) and the coastal ponds of the Languedoc-Roussillon (France). There has been an evolution in such projects, with time, from those modest aims small in area to more ambitious and complex aims and large-scale applications.

The recovery of endangered or threatened species, with special emphasis on waterfowl, comprised the first restorations of Mediterranean coastal wetlands, just as in other types of wetlands. Particular emphasis was placed on habitat restoration critical for species conservation. First morphological restoration (increased flooded surfaces) projects and later re-vegetation. These actions have been very successful, because, among other things, technical difficulties were not very great. However, these first experiences quickly demonstrated that hydrology was the key element to be considered.

Often, habitat restoration is approached considering a pristine situation as the baseline. This implies a natural state, practically free of human interventions, with conditions for the greatest biodiversity possible. This is an unrealistic scenario on which to base restoration. Another typical error is a biased approach weighing ecosystem management to benefit a single species and not biodiversity as a whole.

With respect to the restoration of hydrological functions, most Mediterranean wetlands, suffer from water quality-and/or quantity-related problems. The solution to this type of problem is to control the activities generating it, but projects for water purification and flood control must be sustainable without constant human assistance. The application of the European Union Directives on the quality of water (focused on waste control) and the arrangement of important European financial funds destined to improve environmental quality as a whole has been very useful.

Many actions have been predicated on the development of rules and legal specifications on waste control. Projects dealing with the rehabilitation of polluted water bodies have yet to begin in the Mediterranean basin. Under the existing circumstances, if pollution sources are controlled, most systems should recover via self-purification.

Another type of project has been based on the rehabilitation of the water quality supplies through the assessment of water uses (executing better distribution, application of water saving and recycling techniques by consumers).

Hydrological restoration projects, compared with habitat restoration, are more complex and when aims are set, these are less expensive. Thus, when there is awareness of the degree of water degradation or over-exploitation, it is thought acceptable to return the situation to a condition immediately preceding the degradation. To reconstruct the original hydrological conditions of a wetland is neither practical nor feasible. It must be kept in mind, as Custodio (1995) pointed out, that this is not possible in wetlands with fluctuating water levels (as are most). Neither is a complete restoration of a system to natural conditions in accord with the dynamics of the wetland. Action must be focused on the elimination, as far as possible, of the effects of the hydric budget both in quantity and quality.

Other projects that have frequently been undertaken in the Mediterranean coastal wetlands have focused on shoreline protection. In the past, more attention was paid to protecting the wetland from unwise activities exercised along the shore than to defending the wetland from coastal erosion. There have been “hard” interventions such as detached breakwaters, groins, sea-walls, and jetties, as well as “soft” interventions such as regenerating of beaches and dunes.

The situation mentioned above accurately describes that of different restoration projects. Programs having the restoration of less tangible functions of the wetlands as aims, such as the retention of pollutants and trapping of sediments and toxicants are not yet considered. They are still beyond the wareness, perception and evaluation not only of the general public, but also of many managers.

The recreational value (nature tourism), on the other hand, receives greater emphasis and is considered vital for citizen awareness on the subject of wetland conservation.

A highly relevant aspect, but largely overlooked, is the recovery of the wetland cultural heritage. This recovery is practically impossible because it cannot be repeated and depends on history. What is still possible, however, is the recovery of the traditional uses of the wetland that have long co-existed with natural values and tools of development and that are a part of the ethnological heritage of the peoples of the Mediterranean. This is a subject that has been developing so strongly lately in MedWet/Com that the Technical Session of its 3rd Meeting (2000) has been devoted to it.

Restoration initiatives have had varied success, but valuable experience has been gained. The most frequent cause of failure has been the lack of detailed scientific knowledge on the genesis, evolution, hydrodynamics and functions of the wetland to be restored. Another problem has been the lack of engineering application to wetland restoration. Although there has been major development of the subject in the USA (Odum 1987, Zedler 1988), Mediterranean coastal wetlands have a different functioning regime. Models imported from other biogeographic and hydrodynamic environments are not entirely valid here. American experiences are mainly in temperate areas with abundant rainfall, a tidal regime and great availability of fresh water. This cannot be applied directly to semiarid zones, both coastal and inland, with scarce water resources and bordering a sea lacking major tidal influence.

On the other hand, biological and hydrological functions are most difficult to restore in coastal wetlands. There are many natural factors that affect the hydrologic cycle and many water users and pressures on the land and other resources of the wetland. For these reasons, such interventions have been more modest. Besides, evaluation of mid and long-term effects is necessary. It is too early to proceed with such an evaluation in many of the projects undertaken.

Another important problem was, and is, the monetary cost of restorations, and the time scale or scope at which results are produced (restoring an aquifer can take a very long time). European Union funds have been required to enable the implementation of such initiatives. Few Mediterranean coastal wetlands through the mid 1990’s, benefited from this financial aid, but in the last half of the decade this has grown spectacularly. Most initiatives have been financed with ACE-ACNAT, ENVIREG Program and LIFE funds.

Funds from ACE-ACNAT have financed projects for re-flooding or raising the water table and restoring both water quality and former human activities (Mondain-Monval 1995). In general, the original approach was oriented towards the conservation of waterfowl. These funds have been widely used in France (e.g. rehabilitation of the Marais de Vistre) and Italy.

Restoring water quality was the focus of the next phase. Several of the measures financed have aided wastewater management. For example, the Regulation Medspa for the Mediterranean region financed projects to protect the environment.

The ENVIREG program includes measures to help the less-developed regions of the European Union preserve their environment while promoting economic development. Under these programs, actions to fight pollution, especially in the field of water quality improvement, such as connection to sewage networks, and building of urban or industrial wastewater treatment plants, are contributions to the restoration of coastal wetlands located immediately downstream of wastewater discharges.

Projects related to wetland restoration were funded by the LIFE Program. Their objectives was to improve water quality. They were eligible for funding under the line *"promotion of sustainable development and the quality of the environment"* according to the specifications and terms of this program. Some of them will directly benefit wetlands, such as those dealing with wastewater treatment. Other projects involving wetlands restoration measures were eligible under the line *"protection of habitats and nature"*.

Other legal or financial means are concerned with agro-environmental regulations that accompany the reform of the Common Agricultural Policy (CAP), the "Directive concerning urban waste water treatment" and the "Directive concerning protection of waters against pollution caused by nitrates from agricultural sources". These provide an important means for improving water quality.

The Structural Funds and community initiatives, which aid the poor regions and weakest sectors of the EU, comprise the implementation of conservation or restoration measures in protected areas, especially those concerned with EU directives related to nature conservation. The Cohesion Financial Instrument provides financial contributions in the environmental sphere (particularly concerning quality).

There are no large scale American style restoration projects of coastal wetlands in the Mediterranean currently. Soon there will be projects at Doñana, commissioned after the polluting spill from the Aznalcollar Mine managed by the company Boliden, and the project M.O.S.E. to be implemented in the Laguna di Venezia. Restoration in Doñana must be approached from the purification perspective after the serious incident that occurred in 1998 when a great volume of toxic mud affected the wetland and its zone of influence. In this situation, soils, surface waters and groundwaters must undergo complex remediation actions. It will be possible to evaluate results of these actions in the mid-and long-term.

The Laguna di Venezia has traditionally been a wetland topic in many interventions aiming to stabilize the environment (prevent its natural evolution) and to guarantee maintenance of existing uses (Favero 1992). Throughout the area, highly degrading activities are observed (industrial areas, heavy navigation, and intensive tourist use). The M.O.S.E. project aims to reverse the degradation of the Venice Lagoon and to protect the cities against tides (Gobbi 1995). This polemic project is based on a complex, mobile system of sluice gates to control both the lagoon outlets and the tidal flow, safeguarding the city from extraordinary tides. Some scientific organizations have expressed serious doubts regarding the convenience of controlling the hydrologic regime and the effect it could have on ecological dynamics.

Conclusions

In the Mediterranean basin, coastal wetlands have been subject to many interventions and are still very desirable areas for activities such as intensive agriculture and especially tourism.

Despite the loss of a great part of this type of wetlands, the dominant strategy at this time is conservation. Foreseeing and preventing degradation of the existing wetlands through preventive measures (management and protection policies) must be a priority. There is no experience in large-scale restoration projects, although “Doñana 2005” has already started.

Considerations based on restoration experiences suggest that detailed knowledge of ecological and hydrological processes and their dynamic evolution is necessary to guarantee restoration success. These are ecosystems with a specific functioning regime that do not adhere to oceanic models and where water fluctuation is a determining factor.

The firm commitment of the European Commission to environmental restoration is important. This commitment has been reflected in various financing programs, and these are the main economic support of the restoration of Mediterranean coastal wetlands.

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THE WATERSHED APPROACH FOR RESTORATION OF MEDITERRANEAN RIVERS: THE AXIOS RIVER

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Introduction

Since ancient times rivers have offered invaluable goods and services to humans: they have provided drinking and irrigation water, food for humans and farm animals, game, easy transport routes, nutrients to agroecosystems, energy etc. It is not surprising, therefore, that Mediterranean rivers, notably the Nile, are associated with the development of civilizations. Rivers were looked upon with adoration but also with awe due to the flood damage they often caused to crops and settlements. One may note that the recognition of rivers as major contributors to the preservation of biodiversity is a fairly recent development.

During the previous century man developed the will and the technology to maximize the economic values (in the narrowest sense) of rivers and to minimize flood damage. Thus, rivers were embanked, deepened, diverted and dammed. Riverine forests were clear cut and turned into cultivated fields. Moreover, many rivers were used as inexpensive sinks for industrial and municipal pollutants.

The results of those interventions were the degradation or total destruction of riverine ecosystems (Pyrovetsi 1983) as well as other wetlands which were indirectly influenced by rivers. River water quality was altered not only by alterations to the rivers themselves (e.g. direct input of pollutants, flow interruptions during the dry summer months followed by sea water intrusion) but also to their watersheds (e.g. changes in sediment and nutrient transport due to unwise management of cropland, rangeland and forestland).

Several examples can be drawn from recent history. In both the Ebro river in Spain and the Rhône in France, public works designed to protected riverbanks and flow deviation resulted in degradation and loss of wetlands through either direct land acquisition or modifications to temporal flood dynamics. In Greece, the construction of a dam and excessive use of the Nestos river for irrigation resulted in salt-water intrusion at its mouth and the alteration of delta wetlands. In Egypt, the Aswan high dam prevents the natural flooding cycle of the Nile, thus altering the sediment balance of the delta and reducing agricultural productivity. Although the fertilizing effect of floodwaters is utilized today in some developing countries (Welcomme 1979), most developed countries in the Mediterranean basin use dikes in lowland agricultural areas to keep floodwaters out of fields, and chemical fertilizers are applied to maintain soil productivity. Both actions affect adjacent wetland ecosystems. In the Po river basin in Italy, the extraction of sand and construction of a dam have also reduced sediment transported and the areal extend of the delta.

In addition, river pollution is common throughout the Mediterranean. In most of the cases, non-point source pollution is the main threat for wetlands, but there are also cases where point source pollution is severe including the Po river in Italy, which received industrial heavy-metal pollution, and the Odiel and Tinto river basins in Spain, which have been affected by shipping (Morillo and Gonzalez 1995).

Rivers are a reflection of their drainage basins (Hynes 1975), and they are often adversely affected by watershed management practices such as grazing, road construction, flood control, agricultural and irrigation practices, logging, mining, wastewater management and recreation (Committee on Restoration of Aquatic Ecosystems 1992). The objective of this chapter is to point out the importance of using a river-basin management approach in the restoration of Mediterranean rivers and their wetlands and to present the case study of the Axios river as an example of restoration through wise use of irrigation water.

Constraints in restoring river-dependent ecosystems in the Mediterranean basin

The management of Mediterranean rivers, including restoration and water allocation, is a contemporary issue of great significance. Two or more Mediterranean countries share more than 20 river basins. Although several joint benefits and advantages may arise through cross-border river management policies, effective international river-basin management schemes are lacking, because countries, having initial access to rivers higher in the watershed are afraid of a future water shortage that might limit their political and economic options and therefore maximize utilization of the resources (Gleick 1994). In addition, bilateral agreements for river management are scarce, and in some cases, tensions exist even between supposedly cooperating countries (Lekakis 1998). As a result, the restoration of Mediterranean rivers becomes difficult, and efforts are usually insufficient to address the problem. For example, there is a joint commission between Syria and Jordan for the planning and development of the Yarmuk river basin. However, Syria has constructed several dams that were opposed by Jordan (Gleick 1994).

Many bilateral agreements are general, not stipulating specific avenues of cooperation, and lack the perspective and capacity to produce any real action towards efficient cross-border water management (Lekakis 1998). Such agreements exist for example, for Southern Balkan rivers such as the Strymon, Ardas, Nestos and Evros, which are shared by two or three countries (Bulgaria, Greece and Turkey).

Efforts to restore rivers and related wetland ecosystems are complicated by the hydrologic and sediment regime changes typical of most rivers (Committee on Restoration of Aquatic Ecosystems 1992). Usually the restoration of wetland functions is more feasible, while a return of these wetlands to a prior condition is almost impossible without removing dams, dredged channels and other infrastructure (Henry and Amoros 1995). River-basin management plans have been proposed as a comprehensive and integrated approach to preserve and restore ecosystem health, founded on an ecological basis, and as a means to improve holistic regional-scale management (Johnson et al. 1998).

The watershed approach to river restoration

Changes in land use, cropping patterns, and land management frequently result in altered wetland hydrology and water chemistry, which affect wetland functions (Owen 1999). Managers need improved methods for predicting what levels of these functions are both possible and sustainable based on existing local constraints (Strange et al. 1999). In river restoration, the re-establishment of natural hydrology and a return to original conditions is no longer an option. Actions at watershed level are far more important for the restoration of rivers. Such actions fall into three categories: a) creation of wetland functions, b) modification of land cover and land use and c) management of water resources.

Creation of wetland functions

The creation of wetland functions recently assumed a prominent position in the sustainable management of the rivers in the Mediterranean basin. For example, the construction of a wetland for municipal wastewater treatment near the city of Thessaloniki in Greece stemmed from the need to preserve the water quality of a nearby river (Zalidis et al. 1999). Constructed wetlands are effective for water quality improvement, stabilization of riverbanks and wildlife conservation. Two of the most important wetland functions, which are used to improve water quality entering into the rivers, are nutrient removal/transformation and sediment/toxicant retention.

Constructed wetlands built to perform these two functions are used worldwide for controlling both point and non-point source pollution. They have been applied to the treatment of wastewater from municipalities, industrial plants, landfills, mines and urban and agricultural stormwater runoff. (Kadlec and Knight 1996). The design of such wetlands depends mainly on the types and concentration of pollutants they are designed to treat, the volume of wastewater that they receive, the expected degree of treatment and the environmental conditions in the construction area. In general, wetlands can remove and transform both organic and inorganic materials (including municipal waste, toxic compounds and heavy metals) from inflowing waters (Best 1987, Maltby et al. 1988). In Estarreja, Portugal, a wetland is used to treat the wastewater of a chemical industrial plant (Martins Dias 1999, personal communication).

Nutrient-rich waters flowing into a river may lose some of their nutrient load when passing through wetland vegetation (Peterjohn and Correll 1984, Soranno et al. 1996). Vegetated wetland buffer zones along rivers have been effective for controlling non-point source pollution and restoring the water quality of rivers. Wetland buffer zones can reduce the concentration of nutrients, pesticides and sediments in the surface runoff and thus prevent the degradation of water quality in streams and rivers (Perry and Vanderklein 1996). Potential pollutants carried in stormwater runoff from urban areas are a major component of non-point source pollution. Constructed wetlands are not only able to reduce the concentration of such pollutants, but they also have an added value for urban areas in terms of ecological refugia, aesthetics and environmental education (Livingstone 1989, Ferlow 1993). Thus, the presence and location of a wetland can strongly affect the movement of materials across the landscape and can contribute to water quality improvement in rivers.

The role of constructed wetlands in soil conservation has been identified recently. In accordance with the recent Water Framework Directive of the European Union, which calls for maintaining the quality of water bodies and their adjacent terrestrial ecosystems, the soils of a basin must be maintained in a "good status". The establishment of wetland functions for water quality improvement may prevent soil degradation due to the discharge of untreated wastewater on the soil surface. Wetlands are also used for the improvement of soil quality. The drainage of Lake Karla in Greece resulted in salt water intrusion into the ground water aquifer and the salinization of soils due to irrigation with salt-rich water. The restoration of the wetland and re-establishment of the ground water recharge function will hopefully improve the quality of the ground water. Furthermore, irrigation using water of appropriate quality may leach out excessive salts from soils and contribute to their restoration.

In many cases around the world, wetland plants are used for the phytoremediation of soils. Plants such as *Typha* sp. may remove heavy metals from soil and water (Reddy and Smith 1987), while others (*Colocasia esculenta* and *Ischaemum aristatum*) favor the degradation of organic pollutants and pesticides (Cheng 1999, personal communication).

In addition, wetlands may be constructed in order to provide wildlife habitats and support the foodweb of degraded wetlands. These wetlands are intended to compensate somehow for the rate of loss of natural habitats resulting from agriculture and urban development. They may be built as part of overall environmental enhancement programs or as compensatory mitigation for specific losses of natural wetlands. Of course a constructed wetland can neither completely substitute for the functions of a drained wetland, nor used as an excuse to drain more natural wetlands.

Modifications of land cover and land use in the watershed

There is a tight relationship between land cover/use in a watershed and the quantity and quality of water reaching a river. Streams flowing through forest or rangeland generally carry less sediments and nutrients than those flowing through cropland (other factors being comparable). Differences may also be identified between sub-types of each of the three broad land cover/use types, e.g. between crop species, between species of grazing farm animals etc.

A factor of cardinal importance is the management practices followed when using part of the watershed for crop production. Some practices (e.g. plowing across contours, crop residue burning, fertilizer application on the soil surface, irrigation) are conducive to greater export of nutrients and sediments to rivers and streams (Tsiouris et al. 2002a) than others (e.g. plowing along contours, soil incorporation of crop residues and fertilizers, dry land farming). Irrigation in Mediterranean watersheds is the single most important agricultural practice affecting wetland quality (Gerakis and Kalburtji 1998).

Forest management practices differ also in their effects on rivers. Clear cutting enhances water discharge to rivers that is richer in inorganic sediments, detritus and dissolved nutrients.

Clearly, adopting the best management practices for each land cover/use category of the watershed is the most effective approach to the conservation of wetlands. In most Mediterranean countries, however, this approach is constrained by a lack of data documenting the best practices. The authors have encountered numerous cases whereby many users of watershed resources are willing to change their practices to more sustainable ones, but do not know how. There are also cases where both knowledge and will exist, but the cost of implementing the best management practices exceeds the capacity of the local community.

Industrial and urban activities also affect rivers through either accidental or continuous inputs of pollutants (Tsiouris et al. 2002b). Heavy metals, organic pollutants, nutrients and petroleum by-products have been documented in several Mediterranean rivers such as the Po in Italy, the Ebro in Spain and the Rhône in France. Environmental development planning, the control of point source pollution through the effective treatment of wastewaters and the application of strict environmental regulations may contribute to the conservation and restoration of rivers.

Water resources management

The effective management of water resources in watersheds is of primary importance for river restoration. The extensive use of rivers for agricultural, municipal and industrial purposes has become a threat for numerous wetlands suffering from water deficiency. The change in the water chemistry of several Mediterranean rivers is one of the most important environmental pressures. One major impact, which visibly affects the flora, is the development of high salinity in the delta area during peak irrigation periods. Effective design of irrigation networks, increased irrigation efficiency, and use of less water-demanding crops may minimize water loss (by evapotranspiration) and decrease water demand from rivers. The treatment and reuse of wastewater for irrigation and industrial purposes (e.g. cooling of machinery) are water-saving alternatives to reduce water demand from rivers. In addition, the management of floodwaters through de-synchronization of flooding events, storage of floodwater and gradual release may be a solution for chronic water deficiency in rivers. Furthermore, floodwater may be used for recharging of groundwater aquifers as in the cases of the Shiquma and Nahalei Menashe recharge areas in Israel. Because of high annual evapotranspiration, water storage in aquifers in the Mediterranean basin can minimize the losses of water, reduce pressure from water demand on rivers and contribute to the sustainable management of water resources (Acreman 2000).

Restoration of the Axios river through sustainable management of irrigation water

Site and problem description

The Axios river flows from former Yugoslavia through Greece to the Gulf of Thermaikos. About 90% of the river basin lies in former Yugoslavia and 10% in Greece. In the last 10 km of its course (Northern Greece, Figure 1), the Axios has suffered serious degradation in recent years due to an increase in the salt concentration of its water. Reduced river discharge in summer is mainly due to drought and the diversion of river flow into irrigation networks, leaving no water to reach the delta. The irrigated area (30,000 ha) is divided into 10 irrigation networks, seven of which are surface with concrete-lined channels and three of which are sprinkler systems. A special feature of these networks is the large but fluctuating portion of the area under rice cultivation. Drainage water from irrigation networks, due to land topography, does not return to Axios. It partly drains to the sea through drainage pumping stations, and the remainder drains to the nearby Loudias river (Figure 1).

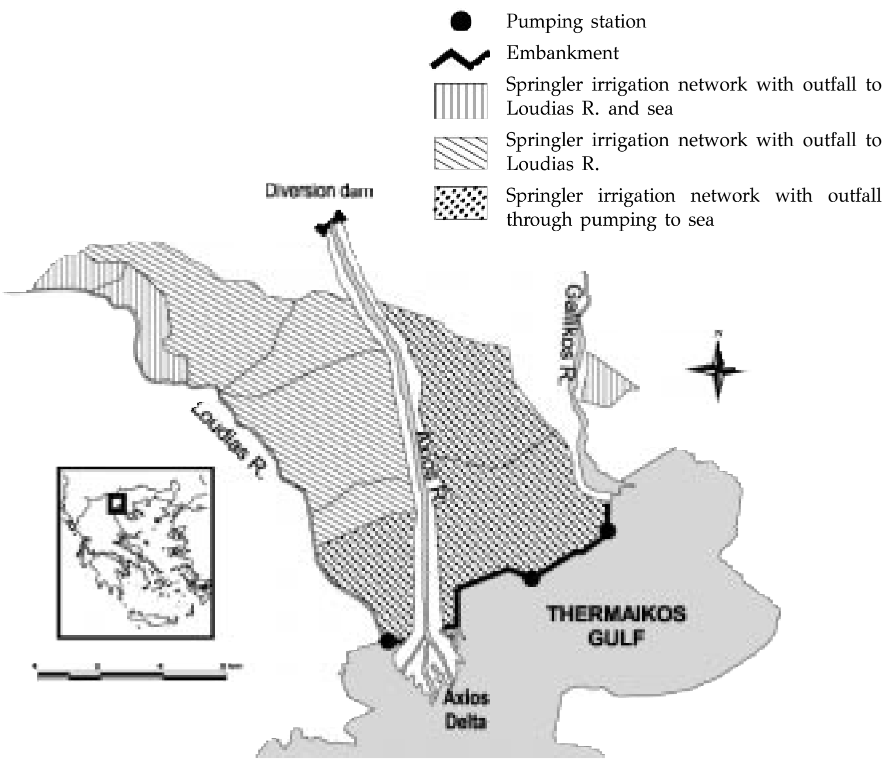

Figure 1. The Axios irrigation networks.

Due to the interruption of the discharge at the river mouth, seawater moves upstream under the influence of the tide. Salinization has an impact on both the flora of the delta and the soil quality of the surrounding area. The differences in salinity along the river, which were revealed by measurements taken at three time intervals after the cessation of the river's natural flow, are presented in Figure 2. It is clear that salinity increases towards the river mouth and that the rate of upstream salinization is a relatively slow process requiring a long interruption in the river flow. In the Axios, as elsewhere, flood control structures channel most of the floodwater directly to the sea and most effluents are deposited in evaporation basins rather than used to support the river delta. Glenn et al. (1996) suggest that in such cases restoration plans should maximize the benefits to wetlands by allowing irrigation return flow to enter deltas. However, the topography of the Axios Delta does not allow irrigation return flow to enter the delta, and for this reason, the restoration plan of the study area does not follow this principle.

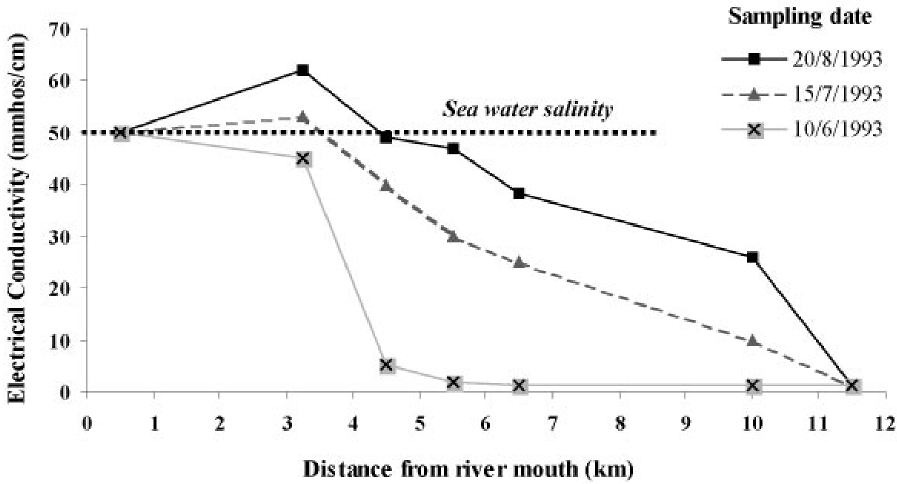

Figure 2. Variation in electrical conductivity of the Axios river, as a function of distance from the river mouth (Dimitriadis 1995).

The prospects of increasing river discharge in the future are very poor, especially considering that rainfall during the last 20 years has declined by 25% in the Balkans and former Yugoslavia is increasing its water consumption of this transboundary river every year. Thus, river discharge will likely remain below the 40 m^3/sec capacity of the irrigation networks during the peak pumping period of July-August.

The practice followed until now was for a fixed supply of water, based on a specific discharge (1.1 liter/sec/ha) estimated during project design, to be delivered to each network regardless of changes in the area covered by each crop type, without taking into account any precipitation during the relevant period. Excess water supply leads to waste and misuse of irrigation water in addition to increasing non-point source pollution (Jensen et al. 1967, Bos and Nugteren 1974). A review of the current literature reveals that while there is considerable information on water management issues (e.g. Barrocu and Puddu 1987, Kulga and Adanali 1990, Lytras and Tsiourtis 1990, Mangion 1994) and irrigation efficiency (e.g. Bos and Nugteren 1974, Butzer 1985), there is an information gap on the water resources management of agroecosystems; such information is needed to mitigate the environmental impact on the adjacent wetlands (Papazafiriou 2000).

Improvement of irrigation efficiency to mitigate salinization of the Axios river

The solution to the problem of Axios river salinization lies in maintaining a specified minimum river flow and in diminishing the duration of interrupted flow through the delta. Since there are few alternative sources of water in the entire area, the only way to keep a constant river flow, especially during droughts, would be to save water by increasing the irrigation efficiency of the networks fed from the river (Zalidis et al. 1997). For this reason, an irrigation management system was proposed (Zalidis 1998) to estimate water quantity that should be supplied to each irrigation network according to its real requirements.

The water resource management system was developed using a Geographic Information System (GIS) and was based on fully distributed real time inputs (climatic data). The system adjusts network irrigation water supply through a feedback mechanism. This mechanism responds to changes that might occur in the exogenous system inputs (i.e. effective rainfall and actual evapotranspiration), and to the difference between the system's target values and calculated values. In this way, irrigation water is used efficiently, thus minimizing the degradation of natural resources (Zalidis 1998). The whole method is based on the following two assumptions: a) field moisture is the same at the beginning and the end of the irrigation period (no net gain in water storage during the irrigation period) and b) ground water level remains approximately constant.

By applying the above irrigation management system, it is expected to reduce the period during which river flow interruption occurs from 50 days to 10 days. This would have a very beneficial effect on the delta because salinization is a slow process, and therefore such a reduction in the period of flow interruption would maintain salinity within acceptable limits, would prevent salinization and would promote the restoration of the river supported ecosystems.

Conclusions

Watershed planning has been proposed as a comprehensive and integrated approach to preserving and restoring wetland ecosystems. The watershed approach is based on an ecological foundation as a means to improve holistic regional scale management. Today, there are opportunities to forage cross-boundary and dynamic bioregional alliances between a wide range of interests that respect and care for particular watersheds. Watershed management, including establishment of constructed wetlands, appropriate modification of the land cover and use, plus sustainable management of the water resources, may provide restoration solutions for several riverine wetlands around the Mediterranean.

The Axios river is a representative restoration challenge, applicable to other Mediterranean rivers feeding irrigation networks, where the management interventions at watershed level may restore the functions and the values of the river delta. The degradation of the river is mainly due to the unsustainable exploitation of river water for irrigation purposes. The reduction of water flow into the sea results in the intrusion of saline water into the river mouth and the degradation of the delta. Using the proposed water resource management system, it may be possible to remedy the environmental impact of agriculture by saving enough fresh-water to maintain the functions of the natural ecosystems. Naturally, the restoration of the Axios and its delta does not depend only on re-instating the flow of water. Several additional measures are needed, such as wardening against illegal hunting, logging, grazing, liquid and solid waste dumping etc.

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WETLAND CONSERVATION AND RESTORATION IN THE MAGHREB

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Introduction

The Maghreb is the name for the north-western part of Africa, derived from the Arabic word for “the west”, and usually taken to mean the three states of Morocco, Algeria and Tunisia, though Mauritania and Libya, which are outside the scope of the present chapter, are sometimes included. The Maghreb enjoys a Mediterranean climate in coastal areas, with considerable winter rainfall and hot dry summers. In the highest parts of the Atlas Mountains rainfall may be as high as 2,000 mm per annum, decreasing to a couple of hundred mm along the northern edge of the Sahara, though amounts of rainfall vary considerably from year to year. As a result there are extensive wetlands: some are freshwater sites in floodplain situations (often called garaet in Arabic); some in closed basins (sebkhet, chott, guelta or daya in Arabic) which often suffer considerable evaporation and dry out in summer or dry winters; there are also a series of coastal wetlands, including lagoons, often connecting with the sea, and in Southern Tunisia the Gulf of Gabès, one of the few major tidal mudflat area in the Mediterranean. A number of general descriptions of Maghreb wetlands exist (e.g. Morgan and Boy 1982), and there is a detailed preliminary inventory of Tunisian wetlands (Hughes et al. 1997).

While the three Maghreb states have joined most of the international conventions and agreements relating to wetland conservation, practical application of their provisions has proved difficult. The prime concern of decision-makers on wetland and water issues has been provision of water supplies for development, and hence construction of large dams has severely affected the original wetlands, particularly in floodplains.

A rapid historical survey of agricultural and water policy in recent years is presented to illustrate this situation. There are as yet practically no wetland restoration projects in the Maghreb, with the exception of Ichkeul in Tunisia, where the problems are so great that it may prove impossible to restore the original wetland values. On the other hand, the construction of small water impoundments in lesser river valleys has, almost by accident, recreated conditions where the biological diversity found in the former major sites can survive. Because of this paucity of restoration projects, the chapter concentrates on the need to include wetland conservation and restoration measures in broader territorial planning.

The present chapter covers the three states of the Maghreb, but concentrates on Tunisia, where the author has been observing wetlands, and in particular their birds, for nearly 40 years.

General wetland situation, climate and relief

Rainfall mostly arrives via depressions of Atlantic origin moving along the Mediterranean coast in winter, and decreases rapidly towards the south and the desert edge. It falls almost exclusively from October to March, often with heavy thundery downfalls in early autumn causing rivers to run in spate. Many of the rivers dry up completely in summer.

In terms of relief, the Maghreb may be divided into:

- a central block along the coast and including the mountains in the hinterland;
- to the west of the central block, the Atlantic coastal plains of Morocco;
- to the east, the lower lying lands of Central and Southern Tunisia;
- to the south, the Sahara.

There are relatively few riverine wetlands, particularly along the north-facing Mediterranean coast of the central block. The North African coast of the Mediterranean is generally mountainous, the Atlas Mountains running roughly parallel with the coastline from Fes in Morocco in the west, almost to Tunis in the east. The coast is characterized by a mainly rocky shore with many cliffs; few large rivers run from the Atlas northwards to the sea.

The only major river valleys and floodplain wetlands are:

In Morocco

- The Moulouya Valley in the west.

In Algeria

- The Arzew wetlands west of Oran.
- The Mitidja plain inland of Algiers.
- The Reghaia coastal wetlands east of Algiers.
- The river Sebaou Valley in Kabylie (which however has a deep valley with little space for riverine wetlands).
- The river Soummam Valley which reaches the sea near Bejaia (but again has a mainly deep-set valley).
- The Plaine des Guerbès and Lake Fetzara west of Annaba.
- The El Kala wetlands east of Annaba.

In Tunisia

- The Oued el Kebir near Tabarka (a relatively small floodplain).
- The Oued Zouara near Nefza (again, a relatively narrow floodplain).
- The Ichkeul complex, where water from half a dozen smaller rivers collect in one shallow basin, before reaching the sea via the much deeper Lake of Bizerta.
- The Medjerdah river which rises in Eastern Algeria and reaches the sea between Bizerta and Tunis.

Of these Atlas riverine floodplain wetlands, the most important by far are the El Kala wetlands in Algeria and Ichkeul in Tunisia. The two rank with the Camargue in France, and Doñana at the mouth of the Guadalquivir in Spain, as the four most important wetlands in the western basin of the Mediterranean.

Ichkeul ("Garaet Ichkeul" in Arabic) is a large freshwater lake (some 8,000 ha in extent) with fringing marshes, situated in a shallow depression, which receives water from two medium sized rivers, the Sedjenane and the Joumine, and half a dozen smaller ones. A dolomitic mountain emerges dramatically from the center of the lake. The water is held up in this depression before moving slowly out to the sea via the much deeper Lake of Bizerta. The lake and mountain have the status of national park, and the whole site has been recognized as a World Heritage site, as well as a Ramsar site and Biosphere reserve.

The El Kala wetlands are a mosaic of varied wetlands, including Lakes Oubeira and Tonga (both Ramsar sites and included in the El Kala national park), plus the unprotected Garaet El Mekhada, and Lac Melah. Garaet Mekhada is a vast freshwater marsh behind the coastal dunes which receives water from two sizeable rivers, the Oued Bou Namoussa and Oued El Kebir; a major dam, the Cheffia, has already been built on the Oued Bou Namoussa; a second is under construction on the Oued El Kebir, though work was halted for a time. The interest of the El Kala wetlands is the great variety of wetlands of different types within a small compass. A little way off, just west of Annaba, is the very important Lake Fetzara, where a number of drainage projects have been carried out, though the lake still floods in wet winters.

At each end of the Atlas, there are broader stretches of flatter land. In the west, in Morocco, a series of major rivers flow through the coastal plain towards the Atlantic (and are hence not really Mediterranean wetlands); these include the Loukkos, the Oum er Rbia, the Sous and the Draa, all of which used to have major freshwater wetlands along their course. In the east, in Tunisia, the Dorsale range (actually the last outlier of the Atlas) runs from south-west to north-east across the country; between the Dorsale and the east-facing coastline of Tunisia is a broad rolling area of plains through which the rivers rising in the Dorsale run. Since the southern section of the Dorsale is near the desert, these rivers run irregularly, but when in spate they create major floodplain wetlands, notably Kelbia, comparable in importance to Ichkeul.

Finally, south of the Atlas, where the desert begins, is a range of wetlands, mostly temporary, fed by rivers running southwards into the Sahara. These are ephemeral and usually salty, depending entirely on local runoff, and exceptional rainfall in the mountains, but occasionally they may be very large. The biggest of them, the Chott Jerid which runs across the border into Algeria, was once a huge lake on the edge of the desert, but is nowadays a salty basin, rarely holding water.

Wetland reclamation

The floodplain wetlands being highly productive, mankind has, for many centuries tried to drain them to improve agricultural production (Carthage was "the granary of Rome", and the cause of the Punic Wars was as much agricultural as political). This tendency has of course increased dramatically in the last two centuries. In colonial times, much effort was devoted to draining the Atlantic plains of Morocco, the Mitidja plains south of Algiers, and the valley of the Medjerdah in Tunisia. Since independence, in 1956 for Morocco and Tunisia and in 1962 for Algeria, the three governments have devoted much attention to increasing agricultural production, in particular by developing irrigated areas. This has resulted in a decrease in freshwater wetlands in particular. Hughes et al. (1997) note that *"28% of Tunisian wetlands have disappeared since 1881, with the highest losses being in the Medjerdah catchment (84%)".* They note that *"the main cause of wetland loss is by drainage"*, but point out that urbanization also accounts for loss.

Because of the nature of the climate, with rainfall occurring mainly in the north, and in quantities varying considerably from one year to another, the three governments have naturally also embarked on programs to improve water supply, whether for agriculture, industry, domestic use or the rapidly developing tourist trade. In Tunisia, this has involved the construction of a series of large dams on the major rivers, and of a series of pipelines and canals making it possible to link up the dam reservoirs, and to transport water around the country, particularly towards the dry central and southern regions.

In Tunisia, some major dams have been built in the mountainous regions near the Algerian frontier (e.g. Barrage Mellegue, completed in 1954, with a capacity of 181 million m^3, Barrage Kasseb completed in 1968, capacity 82 million m^3, Barrage Bou Heurtma, completed in 1979, 117 million m^3). In the 1960's a "Plan Directeur des Eaux du Nord" (Executive Plan for the Waters of the North) was developed by the Tunisian Government, and it has been faithfully carried out in the subsequent decades. The major dams involved were the Sidi Salem on the Medjerdah (capacity 555 million m^3) and dams on two rivers flowing into Ichkeul, the Joumine (capacity 78 million m^3) and Sedjenane.

As noted above, the three Maghreb states are all members of the Ramsar and Biodiversity Conventions. In Tunisia the principal national text relating to wetland conservation is the Forestry Code, administered by the Forestry Department of the Agriculture Ministry, which is responsible for the management of protected areas. This Code, revised by law 20-88 of 1988 includes, under the section on "Hunting and Protection of Game", a chapter on "Protection of Wetlands". Article 224 gives a definition of wetlands, Article 225 states that responsibility for conservation of wild flora and fauna is vested in the Forestry Department, Article 226 forbids the depositing of pollutants in wetlands and states that infilling or drainage of wetlands may only take place after authorization by the Ministry of Agriculture. There is no reference to restoration or management of wetlands.

The 1969 floods in Central and Southern Tunisia

As already noted, rainfall in the Maghreb falls almost exclusively in the winter months, and there are often storms with strong precipitation at the onset of autumn, when after the long dry summer topsoil becomes dry and liable to movement by wind or water. In these conditions there is a likelihood of flooding and of danger to human life and property. An extreme example of flooding occurred in Central and Southern Tunisia in September 1969. Unusual weather conditions (north-easterly winds like the Venetian "bora" projecting cold air from Central and Northern Europe) caused massive rainfall in a very short period. Areas round Sfax, with an annual average rainfall of 300-400 mm, received up to 1,000 mm in 3 days. Heavy rain fell over the Southern Dorsale mountains, causing the rivers to flow in spate and to cause enormous damage. Over 600 people were drowned, while many roads, railways and bridges were washed away.

In the face of this catastrophe, the Tunisian government had to take remedial measures to try to prevent recurrence of such damage. Two of the major results were the construction of dams on two of the major rivers rising in the Dorsale, the Zeroud and the Merguellil, both of which supply Lake Kelbia. The Sidi Saad dam on the Zeroud, completed in 1981, has a massive capacity of 1,200 million m^3, and the El Houareb dam on the Merguellil, completed in the early 1990's, can hold 250 million m^3. The building of these dams was of course entirely consistent with the government's aim to control water resources through the country.

While the 1969 floods were an extreme event (there has been much discussion, because of the lack of historical meteorological records, of whether they represented a once in 100-year or even a once in 500-year event), the natural situation of many Southern Maghreb wetlands is to experience periods of drying out, followed by periods of submersion in years of high rainfall. In terms of biodiversity, they are at their most valuable and productive in these occasional wet winters. In the subsequent spring, they attract large numbers of breeding water birds (flamingos, ducks, stilts, avocets, gulls and terns). There have been few such springs in the last 30 years-only 1970, with a follow-on into 1971, 1974 and 1975, and 1990.

The effect on Lake Kelbia of the Sidi Saad and Merguellil dams (together with the Nebhana dam whose capacity is 87 million m^3 on a smaller tributary) has been enormous. Whereas the lake is a natural depression which used to receive the waters of these rivers (and flowed out to the sea on some occasions in the 1930's), it now receives a drastically reduced input of water and has remained completely dry for much longer periods than before. The only time in recent years when something of the former situation prevailed was in 1990; following heavy rainfall in the previous winter, the dams were nearly full and water was released from the two dams to prevent damage to the dams structures in the case of further heavy rain.

Paradoxically, the situation in the 1990's has been extremely poor from a rainfall point of view. Rainfall in the Southern Dorsale has been particularly poor, and in summer 2001, the Merguellil dam had been standing totally dry for a period of 4 years.

Ichkeul

Because of its pre-eminence in biodiversity terms and its recognition under a series of international conventions, Ichkeul has attracted greater interest and attention, both at government and at specialist level, than other Tunisian wetlands.

In Roman times (Hollis 1977) the lake and marshes covered a much larger area than today. Much of the wetland was reclaimed for agriculture in the colonial period. Even so, an area of over 12,000 ha survived and was declared a national park in 1980. Of this the lake made up 9,000 ha, the marshes 1,300 and the rest was formed by the mountain. The basin received an annual input of some 300 million m^3 of water from the six rivers. The depression retained this water in winter, gradually releasing it to the Lake of Bizerta and ultimately to the Mediterranean via the outflow Oued Tindja. In summer, there was reverse flow of brackish/salt water from the sea into Ichkeul (this inflow of salt water probably increased over the last century since the deepening of the channel between the Lake of Bizerta and the sea for naval and shipping purposes).

The presence of a large body of fresh water in the lake and marshes encouraged the development of rich plant and invertebrate communities. The lake held large quantities of the plant *Potamogeton pectinatus*, it was fringed by a broad belt of reed *Phragmites australis* and the marshes were dominated by stands of *Scirpus*. These plants, and the invertebrates associated with them, attracted large numbers (on average a quarter of a million) of wintering waterbirds. The most numerous were: on the lake itself, diving Pnochard *Aythya ferina*; in the shallows Wigneon *Anas penelope* and Coot *Fulica atra*; and in the marshes Greylag Goose *Anser anser*. Furthermore the mountain was a refuge for many birds, notably raptors which preyed on the birds of the marshes below. During the summer, the lake and marshes supported populations of typical Mediterranean breeding species, including the threatened White-headed Duck *Oxyura leucocephala* and Marbled Duck *Anas angustirostris*. During spring and autumn, the site was a major stopover point for bird migrants moving between Europe and Africa. The lake was also the center of a rich fishery production, essentially of fish which moved into the lake to spawn, and were caught in traps at the outlet when they returned to the sea.

Naturally, the governmental plan for the waters of the north was concerned with the use of this major reservoir of freshwater for economic development purposes, rather than with the biodiversity value. Thus, the plan envisaged the establishment –upstream of the lake, and outside the area of the future national park– of six dams, two of them very large, four of them much smaller. It did not envisage reservation of any water for the ecological requirements of the wetland. Thus when the national park was created (under the administration of the same agriculture ministry which was planning the dams) it was already known that there would be a severe reduction in the inflow of fresh water to the lake, and a resultant increase in evaporation and in backflow of salt water from the sea). An additional difficulty for the national park was the existence, all around the boundaries of the park in the Plaine de Mateur, of intensively cultivated arable land. A project to develop agriculture by improvement of agricultural drainage was put into action in the 1980's and 1990's. Any water which collected on this land was evicted into the park (by yet another department of the same Ministry of Agriculture) through drainage ditches dug in the park itself, which had the effect of draining the marshes.

The likely effects of these plans on the biodiversity of the park were recognized in the 1980's. The Ministry of Agriculture, with financial support from the European Commission and expert input from a number of international wetland experts, carried out studies to investigate their effects and possible remedial measures (e.g. Hollis 1986). It was clear that the reduction of freshwater inflow would adversely affect the vegetation and the waterbirds dependent upon it. It would also affect the fisheries. Among the solutions proposed were: the installation of a sluice on Oued Tindja at the outlet to the Lake of Bizerta to control inflow of salt water; reduction of the surface of the freshwater area by construction of bunds, some containing freshwater, some containing salt water; and releases of waters from the upstream dams. The Tindja sluice was in fact built in the early 1990's (though with considerable difficulty due to the soft terrain). The reduction of the surface option was not retained because of the cost of building the bunds. No measures for release of water were envisaged at that time.

Two of the dams came into operation in the 1980's, the small Ghezala dam (with a capacity of 10 million m^3) in 1984, and the much larger Joumine dam (with a capacity of 78 million m^3) in 1986. The Joumine Marshes were seriously affected by the digging of a drainage ditch in 1981 from the Plaine de Mateur through the marshes in the park to the lake.

In November 1990, a major international symposium on Ichkeul was organized by the Government of Tunisia, with input from the newly established Ministry of the Environment and Territorial Planning. This symposium was important as the first major attempt at the very highest level to find a compromise between economic development and conservation of biodiversity, which were likely to be even more severely affected by the construction of the largest dam, on the Oued Sedjenane in the early 1990's. The symposium decided that through-going studies were needed to define the exact measures needed for the safeguard of Ichkeul.

These studies were carried out in the next 5 years, and resulted in one of the most complete studies of all ecological and economic aspects of a Mediterranean wetland ever carried out (Ministère de l'Environnement et de l'Aménagement du Territoire, 1996). It is worth noting in passing that it is most regrettable that a summary of the scientific results and proposed management measure has never been published in a scientific journal. The study recognized that there had been "*a catastrophic collapse of the ecosystems*", and it recognized that the second half of the 1990's (with the Sedjenane dam being filled, and before remedial measures were in place) would be a difficult period for management of the site. It noted the decision taken by the government to provide water to Ichkeul through releases from a new dam (the Sidi Barrak on another catchment, that of the Oued Zouara), which would be pumped to Ichkeul via the Sedjenane dam. It emphasized the need for careful operation of the sluice on the outflow at Oued Tindja.

What the authors of the study could not forecast was, of course, the weather in the 1990's. Unfortunately, there have been two periods of poor winter rainfall even in the normally wet north of the country. The first coincided with the filling of the Sedjenane dam in the early 1990's; the second with the end of the 1990's, before the entry in service of the Sidi Barrak dam. The situation of the wetland, already affected by a shortfall of fresh water inflow in the 1980's, deteriorated very rapidly during the early 1990's. The three major plant groups proved unable to tolerate such a rise in salinity: the *Potamogeton* growing in the water declined drastically, the belt of reeds round the water's edge died down and has to all intents and purposes disappeared, the *Scirpus* in the marshes died out. As a result, the large wintering populations of waterbirds crashed to a tiny fraction of their previous numbers, with those that feed on freshwater plants like Greylag Geese and Coot disappearing almost entirely and those that can tolerate some salt appearing in much smaller numbers. The breeding birds also disappeared almost entirely because of the loss of the vegetation in which they nested. The increased salinity has also led to a considerable decline in fish catches.

In the years since the filling of the Sedjenane dam the situation has not improved: thus in winter 1995/96 rainfall was above average at 860 mm, 1996/97 was desperately dry with only 355 mm, 1997/98 again above average with 789 mm, 1998/99 average with 578 mm, but 1999/2000 and 2000/01 both well below average. The operation rules for the Oued Tindja sluice calls for it to be closed from late spring until early autumn to retain fresh water, if –and only if– there has been adequate water in the previous winter. If the sluice is closed when water in the lake has a high salt content, this will only encourage evaporation and salinization of the water. What has in fact happened is that the sluice was closed in summer 1996 after good rain in the previous winter, but had to be left open in 1997, because the salinity of the lake had risen to 70 g/liter (i.e. twice the salinity of sea water; the inflow of sea water at 37 g/liter was actually diluting the salinity of the lake). After the relatively wet winters of 1997/98 and 1998/99, the sluice could be closed during the summers of 1998 and 1999, allowing salinity to drop somewhat. But with the inadequate rainfall of the last two winters, it has not been possible to close the sluice, and the salinity is once more on the increase.

As yet, the building work for transfer of water from Sidi Barrak to Ichkeul have not been completed, though they are reported to be nearly ready. A definite decision has been made by the Tunisian Government to earmark water for Ichkeul's ecological requirements –a major step for a dry developing country– though the exact amount has not yet been fixed, nor has it been confirmed that such release will go ahead unconditionally (i.e. even in years of poor rainfall, when water is in short supply for other priority activities). Furthermore, it appears that plans for the three remaining dams on inflow rivers, on the Tine, the Melah and the Douimis are still intended to be carried out.

The situation at present is thus that the Tunisian authorities still intend to carry out the plans for restoration of the Ichkeul wetland, the major measures being provision of fresh water from the new Sidi Barrak dam, and operation of the Oued Tindja sluice.

The plans envisage restoration of low salinity levels, leading to the reconstitution of the vegetation and bird populations. It must be hoped that these plans are successful, but the challenge will be very great: the degradation of the original ecosystems has already been so great that they may be past reconstitution; building of the three remaining planned dams will undoubtedly have a negative effect on the wetland's water balance; decisions on the amount of water to be released and on the conditions of release have still to be taken; and as yet there is no adequate administrative structure to organize an integrated management structure involving all stakeholders.

The parallels with Azraq in Jordan, where there are similar conflicts over conservation of biodiversity and water supplies, are obvious. Ichkeul and Azraq are certainly two key case studies for conservation and wise use of wetland resources in the Mediterranean region.

El Kala

Information on the El Kala wetlands in Algeria is far less complete than that on Ichkeul, and it is important to emphasize that the comments below relate to contacts made some years ago.

The two sites are close together (only 150 km as the Greylag Goose flies), comparable in ecological importance, and face many of the same problems, so comparisons are instructive. As indicated above, the El Kala national park includes only two of the major wetlands of the El Kala complex, Tonga and Oubeira. Much of the park has been designated to conserve the magnificent pine, cork oak and alder forests (including some peat bogs). It would be highly desirable for the protected area to be extended to include other sites such as the Lac des Oiseaux, and above all Garaet Mekhada. Though some 50 km away, the Lac Fetzara complex is also worthy of habitat conservation measures, either as part of the El Kala complex or as a separate entity.

Like Ichkeul, the El Kala wetlands have derived their international recognition because of the large numbers of birds that occur there, both wintering waterbirds, and an array (probably more important than at Ichkeul) of breeding birds including threatened ducks such as the White-headed Duck, Marbled Duck and Ferruginous Duck *Aythya nyroca*, and raptors. Already before the beginning of dam-building operations around Ichkeul, there was clear evidence of movement of birds to Ichkeul and El Kala, particularly of wintering Greylag Geese. Individual birds of this species (marked on the breeding grounds with individualized coded neck-collars, legible by telescope) have been recorded at both sites in the same winter (Skinner and Smart 1984). It is probable that many of the birds that wintered at Ichkeul before the degradation of ecological conditions are nowadays wintering in north-east Algeria.

Some of the same conflicts of resource use that occur at Ichkeul also arise at El Kala. One large dam, the Cheffia dam, has already been built on one of the rivers that supply the Garaet Mekhada, the Bou Namoussa. A second was in construction in the late 1980’s on the Oued El Kebir which supplies both Oubeira and the Garaet Mekhada, though construction was suspended, at least for a while in the early 1990’s. Water levels in the area have been affected by extensive digging of wells in the sandy soil, so much so that Oubeira has almost dried up in the past. Furthermore, pressure of hunting is greater at El Kala than at Ichkeul, and it is likely that hunting disturbance causes the birds to move back and forth. The coastline shore between Annaba and Cap Rosa is a long, magnificent beach, with extensive coastal dunes holding back the waters of the rivers flowing from inland. The beach is already extensively used by local tourists, and new roads and facilities are constantly being constructed.

It is clear that an integrated planning infrastructure, giving weight to both sustainable economic development and to conservation of the very rich local biodiversity, is needed.

Coastal wetlands

Recent developments at the Lake of Tunis also illustrate the need for greater attention to biodiversity conservation in government planning measures. The Lake of Tunis is a large coastal lagoon (of some 6,000 ha), situated between the capital and the Gulf of Tunis. It is bisected by the main road from Tunis to Halk el Oued on the coastal sandbar which also accommodates Carthage. The lake had originally high biodiversity, holding large numbers of wintering and passage waterbirds, notably in the Megrine Saltpans on the southern side. Few capital cities had such a range of natural values on their doorstep.

Until the 1970’s, the lake received most of the waste and storm water from the city, and was becoming increasingly polluted. As a result, a major clean up operation was carried out, with the construction in the 1970’s and 1980’s of a completely new sewerage system (by which waste water was no longer released into the lake but diverted into settling ponds) and measures taken to improve water circulation within the lake.

During the 1990’s, much of the northern sector of the lake was reclaimed for building office and living accommodation. The original biodiversity of this area has almost completely disappeared, despite the creation of a nature reserve on the Island of Chikly.

Recently, a similar fate has befallen the southern sector of the lake. Infilling of the greater part (including the former saltpans) has taken place, leaving a stone-lined basin which will no doubt one day serve as a marina for pleasure craft. It is planned to construct similar office and residential facilities on the reclaimed land. At a very late stage, arrangements were made to establish a very small nature area (of a couple of hectares).

Here too, it is clear that the planning measures gave much greater weight to the need for housing and office space than to conservation of biodiversity. It seems a pity that a greater effort was not made to set aside sections of the original habitat as a nature reserve, which in the long term would surely have had amenity value for the citizens of Tunis as well as attractive qualities for tourists.

A more positive attitude to wetland conservation is apparent in the Cape Bon peninsula, where measures are being taken to conserve some coastal wetlands, notably the Korba Lagoons, under a Global Environmental Facility (GEF) project on Mediterranean coastal wetlands.

Small irrigation facilities

Paradoxically, some freshwater wetlands of high biodiversity have been created in Tunisia in recent years, almost by accident. In addition to building major dams whose aim is to provide water on a grand scale for irrigation or the national water grid, the Tunisian Government has implemented a policy of constructing small dams in minor river valleys, to provide water for irrigation. These are built mainly in the north, in areas where annual precipitation is higher, and typically consist of a stone dam across a valley, retaining water which is then extracted in summer by local farmers. Such dams are of particular importance in the Cape Bon Peninsula, the extension of the Dorsale range into extreme North-East Tunisia, pointing across to Sicily. This area once had a major natural freshwater wetland, the Garaet El Haouaria, which was drained to provide agricultural land before independence; this site, situated at the tip of the peninsula, was on a major bird migration route between Northern Africa and Europe, and once held large numbers of passage and wintering waterbirds. The small artificial lakes now mimic the values of this former site, and indeed have become the major sites in Tunisia for those bird species for which Ichkeul was formerly famous.

Most of these artificial water bodies are of recent creation: one of the older ones, Barrage Chiba (maximum capacity 8 million m^3) was built in 1963, Barrage Mlaabi (capacity 3 million m^3) in 1987, Barrage Lebna (capacity 24 million m^3) in 1986, and Barrage El Khatf only in the last few years. Several other small dams have been built, mainly in Cape Bon, but also (e.g. Barrage Bessbessia) in other parts of Northern Tunisia, and there are plans for additional dams of this type.

Most of these sites have extensive floating vegetation (probably *Zannichellia*) which provides food and also supports grebe nests. Some, notably Lebna, have extensive stands of *Phragmites* and *Typha* at the inflow end opposite the dam wall. They currently hold appreciable numbers of wintering waterbirds, not on the same scale as Ichkeul, but of the order of several tens of thousands of birds. In summer they provide habitat for the bulk of the Tunisian breeding population of two threatened ducks, the White-headed Duck and Marbled Duck, and molting areas for another endangered duck, the Ferruginous Duck. A host of other non-threatened species also occur, notably Purple Gallinule *Porphyrio porphyrio* (once numerous at Ichkeul, but now disappeared). Being sited on a major migration route, they are an important stopover point for any migrant bird dependent on freshwater areas, including herons, storks, spoonbills, waders, gulls and terns. These sites are now without question the most important freshwater wetlands in Tunisia from a biodiversity point of view.

Conclusions

It is thus apparent that there have been significant losses of wetlands (particularly freshwater sites) in the Maghreb in recent years, and that these losses are inevitable in the context of the countries' economic development needs. Wetland restoration projects are therefore few and far between, with the notable exception of Ichkeul, where the chances of success are hanging in the balance.

Nowadays, however, there is a growing interest in biodiversity conservation and, through GEF, funds to finance major projects. The Tunisian Government's decision to recognize formally in water planning measures that a supply of water should be guaranteed to Ichkeul, is a major development. It is to be hoped that concern for biodiversity conservation will in future play a larger role in the planning process at the highest government level. Inventories of sites of major interest (such as the lists of Important Bird Areas, recently developed for each country by the national partners of BirdLife International; or the Tunisian wetland inventory) should provide an important reference and guide in this respect.

There is a need to develop national databases on wetland topics: these should be developed by independent scientific bodies, whether research institutes, universities or non-government organizations, who can provide independent advice to government policy and decision makers. The government bodies which manage protected areas also need strengthening, so that they can draw up and apply effective integrated management plans. The small artificial wetlands which have been created in the last few years, and which have become the main repository for wetland biodiversity, have as yet no protected status and urgently require conservation action.

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RESTORATION OF COASTAL MEDITERRANEAN MARSHES ON FORMER AGRICULTURAL POLDERS: THE VISTRE MARSHES (GARD), FRANCE

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Introduction

Wetland restoration is an important challenge in the Mediterranean basin, both from the point of view of land management and land-use planning and also for nature conservation. Agricultural land abandonment and a more comprehensive approach to land management may in the future favor the development of such projects but their implementation is hindered by the complexity of ecosystem functioning and the lack of scientific and technical references (Kusler and Kentula 1990, Larson 1990). Ecosystem restoration is at the interface between research and application.

In Southern France a series of catastrophic floods has recently increased people's awareness on the consequences of the embankment of the floodplain of rivers. Changes in European agricultural policy and the decrease in income from agriculture and especially ricefields (Barbier and Mouret 1992, Mathevet 2000) provide an opportunity for the restoration of wetlands (Mesléard et al. 1999). An attempt was made to restore a 130 ha former agricultural polder (Figure 1), with associated monitoring to enhance the understanding of the mechanisms responsible for the dynamics of the ecosystem. The term restoration is not used here in the strict sense (Aronson et al. 1993), the reinstatement of the original ecosystem as it was in its natural state being impossible because of the age and extent of hydraulic alterations in the region since the 11th century and intensified in the 19th century. Over centuries the progressive embankment of the river floodplain led to increasing flooding stress in nearby urban areas. The surface area of the project was not sufficient to solve the flood problems. Rather, the objective of the restoration was to establish a demonstration project for re-creating wetlands in the floodplain of the Vistre river.

The area was drained for polderization and agricultural use in the 60's and cultivated until 1993. A remnant of non cultivated land within the area suggests that agricultural practices led to the oxidization of 0.30-0.50 m of organic sediment leading to lower altitude of land surface.

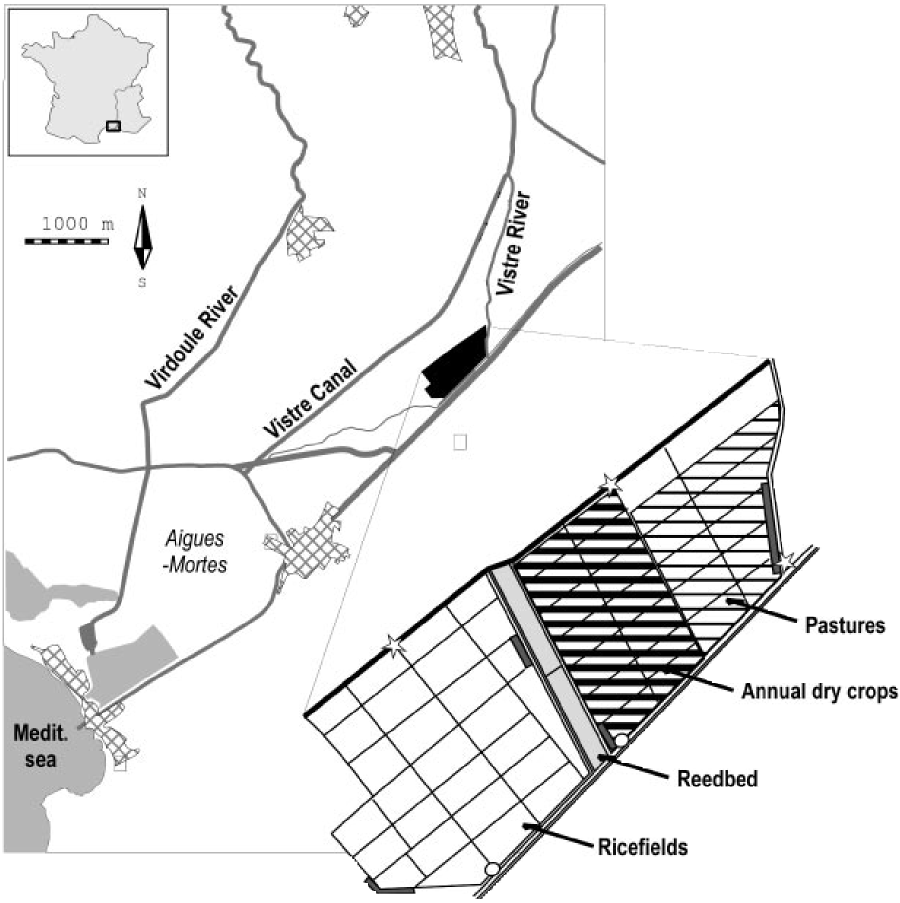

Figure 1. Location of study area and land-uses in 1995. The spillways (3) are shown as black lines along the surrounding dykes, the continuous water level recorders by stars, and the flap sluices by circles.

We attempted to start the ecosystem again on a spontaneous successional path (Aronson et al. 1993) requiring little human intervention after initial management and subject to fluctuations typical of the Mediterranean climate. The main functions desired for the rehabilitated wetland were flood retention, an increased biodiversity through creation of feeding and breeding habitats for waterbirds, and trapping of nutrients and sediments. The restored wetland was also to be used for extensive cattle grazing, reed cutting for thatch and waterfowl hunting. System desiccation in summer was considered to be essential to optimize storage capacity during autumnal floods, to favor development of the target vegetation community and to promote mineralization of organic matter and denitrification.

Predictions of ecosystem dynamics were based on the following hypotheses. Vegetation dynamics was assumed to be controlled by physical environmental factors (Van der Valk 1981, Grace and Pugesek 1997), mainly the hydrological regime and salinity, acting on those plant species already present. The seed bank and the inflow of propagules were assumed to be non-limiting. The outcome of the competition between species is predicted by maximum plant size achieved (Mitchley and Grubb 1986, Keddy et al. 1994). The flooding regime and vegetation structure were assumed to be the main factors controlling habitat use by animals (Kusler and Kentula 1990, Palmer et al. 1997).

Initial study

The restoration approach was determined after an initial study based on 8 years of twice-daily water-level records in the navigation canal 5 km downstream of the site and mean daily discharge of an upstream station on the Vistre canal starting from 1970 onward. A simple hydraulic model was used to reconstruct water level and discharge in the Vistre river from discharge data of the Vistre canal. Data for 2 average-years and 2 extreme-years (dry and wet) were used to calculate water depths and budgets for 10-day periods of the volumes of water entering, stored and exiting the polders for different spillway levels on the dikes. The spillway level on the dikes was determined as a compromise between the frequency of overtopping by the river and the volume stored (and hence the time needed to evaporate to dryness in summer) so as to maximize the length of the dry period. It was set at 0.40 m ASL (Figure 1). The dikes were lowered in the summer of 1995 along a length of 150 m at 2 places in each polder. The initial study also included a study of the soils, vegetation, bird, fauna and the history of land-use.

Monitoring

Water level recorders were installed on the river and in the two former polders. In 1998 and 1999, water samples were collected twice monthly as well as during floods using an automatic sampler. Water conductivity, concentrations of suspended solids (SS) and its organic fraction (OSS), nitrate, ammonium, Kjeldahl-N, orthophosphate and total phosphorus were measured. The chemical analyses were carried out according to the French norms for water analysis (AFNOR): ortho-phosphate (and total phosphorus after acid hydrolysis) using molybdate-antimony, ammonia using indophenol, total nitrogen using the Kjeldhal method. Water samples were filtered (GF/C filters). The filters were dried at 105^{0}C to obtain total suspended material

(TSM), then passed to an oven at 500°C to derive the organic suspended material (OSM). Vegetation was mapped in 1996 using semi-quantitative measurements on 40 to 100 m^2 quadrats in each former agricultural field. Special protocols were developed for monitoring species playing a particular role in the habitat structure (reedbeds, trees), or because of their potential threat (e.g. the exotic species: *Ludwigia peploides*) or conservation value (*Leucojum aestivum*). Fish and macro-crustaceans were surveyed on two occasions (1997 and 1999) using electric fishing, gill nets and seine nets. Wintering and passage birds were counted twice monthly from September to March. Breeding tree-nesting herons were counted annually by visits to colonies. The quality of the site for feeding was estimated from bird counts and by the capture success rate in the case of egrets.

Results and discussion

Hydrology

When below 0.50 m ASL, the level of the Vistre river, at its confluence with the navigation canal, is controlled by the sea, which is almost tideless, the level of the Mediterranean sea itself depending on wind direction and force and atmospheric pressure. Periods when the water level exceeded that of the spillway, 0.40 m, were associated with discharges in the Vistre greater than 10 m^3/sec in only 35% of cases. Rainfall in the catchment therefore only had a major control on the water level during major flood events.

An analysis of the 5 years of water-level monitoring in the Vistre river showed that the frequency of water entries in the marsh (level higher than 0.40 m ASL) had been underestimated in the initial study. The calculations used mean daily values calculated from values recorded at a fixed time, which did not properly take into account extreme events. In 1999, the threshold was exceeded 45 times by the daily mean, but 86 times by the daily maximum (Table 1). The threshold only needs to be exceeded by 5 cm for 12 hours to fill the marsh completely. New, more precise data on the water level regime in the Vistre suggested that a spillway at a level of 0.50 m (Table 1) would have decreased the frequency of water entering by half and would therefore have favored desiccation.

The first few years (Figure 2) were characterized by major floods. Fewer than 80% of fields dried out in summer as a result of evapotranspiration and the installation of the flap sluices in 1996. This was because: 1) spillways were at a level lower than recommended because of poor quality construction and erosion by livestock trampling, and 2) unauthorized manipulations of sluices flooded the marshes in summer. The data suggest that if these problems had not occurred, drying-out would have been more complete and would have taken place earlier. This was finally achieved in 1999 and 2000.

Table 1. Flood characteristics calculated for 2 levels of the spillway: number of days water enter, number of hydrologic events when water enters, maximum duration of water entries and mean duration of water entries in the wetland.

Spillway level	Period	Days	Events	Max. duration	Mean duration
H = 40	Sept.-Nov.	127	27	19	4.7
	Dec.-Feb.	121	13	42	9.3
	Mar.-May	35	12	6	2.9
	June-Aug.	20	7	5	2.6
H = 50	Sept.-Nov.	59	12	14	4.9
	Dec.-Feb.	33	11	33	6.7
	Mar.-May	11	6	3	1.8
	June-Aug.	1	1	1	1.0

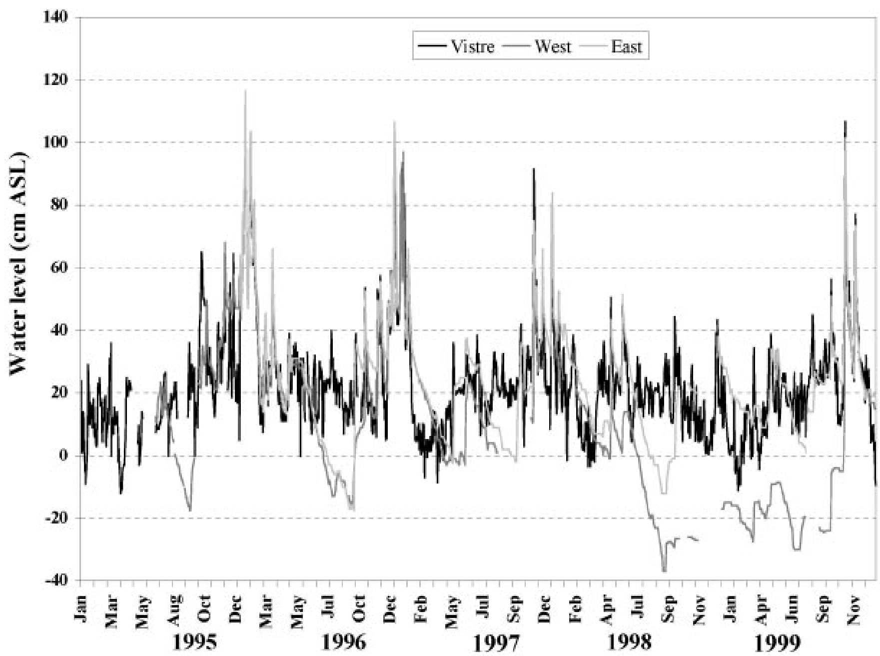

Figure 2. Mean daily water levels in the Vistre river and the eastern and western sectors of the Vistre Marshes from January 1995 to December 1999.

In 1998 and 1999, high sea levels without river floods promoted entry of brackish water to the system. Salt deposition approached 3000 tonnes in the eastern sector and 4500 tonnes in the west. Resulting water salinities in summer 1999 reached 8 ppt in the eastern sector and 36 ppt in the west. During the floods of November 1999, this salt was washed out, and the stocks were reduced to less than 300 and 200 tonnes respectively. The differences in salinity and water level between the two sectors resulted from a disparity in spillway heights. The eastern sector had more frequent entry of water and dried out more slowly, but its salt was washed out more readily.

The water quality of the Vistre river was considered poor, with high concentrations of ammonia-nitrogen (4 mg/l in March 1999) and orthophosphate (2.8 mg/l in June 1999). During floods (from 18 October to 4 November 1999) the chemical budgets were estimated using mean daily concentration of nutrients and calculated water losses through the flap sluices. Calculations for the eastern sector suggested that there was a net export of nutrients and SS as the flood subsided that was greater than or equal to imports. This is attributed to wind and wave action maintaining high SS concentrations in the water. The marsh certainly plays a role in improving water quality by controlling exchanges between the water and sediment and by denitrification. It does not, however, seem to act as a SS and nutrient sink.

Vegetation

Before the start of the project the vegetation of the fields was very heterogeneous, varying with land use (rice-fields/dry annual crops/permanent grassland) and land elevation. After lowering the dikes two trends were observed: a change from terrestrial and wetland to hydrophytic vegetation, and from a plant community including only salt-intolerant to one including salt-tolerant species. This succession took place in two stages. As soon as the dikes were lowered, the flooding exerted a strong selection on the species present, with the disappearance of those that could not tolerate submersion (Table 2). From 1996 onwards, the dominant species were submerged and emergent species included *Potamogeton pectinatus* (17.8% of quadrats), *Chara* sp. (8.6%), *Phragmites australis* (8.0%), *Ceratophyllum demersum* (3.7%), *Paspalum paspalodes* (6.0%), *Potamogeton pusillus* (4.8%) and *Zannichellia pedunculata* (4.4%). There was little similarity between the vegetation present in 1996 and the seed bank, because excessive submersion did not provide suitable germination conditions for species such as *Aster squamatus*, *Polypogon monspeliensis* and *Samolus valerandi*. Most of the hydrophytes did not originate from the seed bank as demonstrated for *P. pectinatus*, *Z. pedunculata* and *C. demersum*, which were abundant in the drainage ditches surrounding the fields and likely colonized from there. Some species, such as *Ludwigia peploides* and *Najas marina* in the eastern sector, seem to have been introduced exclusively by the floods. Dominance of the vegetation by hydrophytes was also reflected in a pronounced decrease in both the proportion of terrestrial species in the plant community and the seed bank (Table 3).

Table 2. Dominant species in the vegetation in 1994, 1996 and 1999 in the different former land use types.

Species	Ricefields			Dry annual crops			Grassland		
	1994	1996	1999	1994	1996	1999	1994	1996	1999
Potentilla reptans							■		
Festuca arundinacea							■		
Aster tripolium				■			■		
Echinochloa crus-galli	■								
Polypogon monspeliensis				■					
Scirpus lacustris				■					
Typha angustifolia				■					
Paspalum paspalodes	■	■		■	■				
Phragmites australis	■	■		■	■				■
Scirpus maritimus	■	■		■	■				■
Chara sp.		■						■	
Ceratophyllum demersum					■				
Zannichellia pedunculata		■			■			■	■
Potamogeton pectinatus		■	■			■			■
Myriophyllum spicatum									■
Ranunculus baudotii									■
Tamarix gallica									■

Table 3. Comparison of the proportions of terrestrial species in the seed bank in 1996 and 1999 (in % of the total number of species and the total number of seeds) for the 3 different former land-uses.

	Ricefields		Dry annual crops		Grassland	
Years	1996	1999	1996	1999	1996	1999
Species number (%)	19.3	15.7	34.7	9.1	36.3	17.2
Seed pool (%)	0.8	8	39.4	2.3	45.8	14.2

In a second stage of the restoration, the entry of brackish water led to dominance by salt-tolerant species. In the eastern sector only 6 species of hydrophytes were recorded in 1999 compared to 17 in 1997. The area of the most salt-tolerant species (*P. pectinatus*, Van Wijck et al. 1994) also increased, whereas *Z. pedunculata* and *Chara* sp., which were dominant in 1997, almost disappeared. The overall decrease in the cover of hydrophytes can be explained by the high SS concentrations resulting from inputs from the Vistre river and the low cohesion of the sediment after several years with brief or no desiccation. The increase in salinity almost eliminated the exotic species, *Ludwigia peploides*.

Reeds (*P. australis*) declined at elevations below -0.20 m and remained stable at higher levels. High water levels and associated severe anoxia resulting from poor water quality in the Vistre river as well as grazing by coypu (*Myocastor coypus*), probably contributed to the reed decline (Ostendorp 1989, Hellings and Gallagher 1992). Regeneration from seed was recorded only on the two plots of highest elevation and subjected to only brief flooding.

Fraxinus angustifolia extended its distribution on both the surrounding dikes and the system of internal dikes on land that was never flooded. *Tamarix gallica* started to colonize the highest fields in the eastern sector. Development of woody vegetation will play an important role not only for the avian fauna but also for the functioning of the marsh by reducing the effects of wind, including sediment resuspension.

All the species dominating the structure of the plant community were present immediately, and controlled mostly by physical environmental factors. The project objectives regarding the vegetation structure in the marsh were not fulfilled because of the hydrological conditions that prevailed in the first few years after manipulation. The “reedbed” objective was partly fulfilled on higher land but is not likely to be feasible in areas with the lowest elevation unless the hydrological objectives (drying-out in summer) are achieved. A coypu control program could favour the development of reedbeds by removing a major exotic grazer.

Fish and crayfish

The objective of the study of fish and crayfish communities was to determine species composition and, reproductive success on the site and to assess their availability as prey for fish-eating birds. A total of 17 species of fish and 2 species of crayfish were recorded (Table 4). These species are characteristic of the lower reaches of both rivers and standing or slow-flowing waters. As the site connects to the sea, migratory and marine euryhaline species, frequently encountered in canals and seasonally-flooded marshes in the Camargue (Crivelli 1981), were present. Nine species of fish and both crayfish species were non-indigenous (Table 4). All have been established in France for a long time (Rosecchi et al. 1997).

Table 4. Species of fish and crayfish recorded on the site, I: Indigenous, NA: North America, As: Asia, Ec: Central Europe.

	Species	Origin	Major canals	Minor canals	Plots
Fish	*Anguilla anguilla*	I		X	X
	Micropterus salmoides	NA	X	X	
	Blicca bjoerkna	I	X		
	Carassius auratus	As	X	X	X
	Cyprinus carpio	As	X	X	X
	Gasterosteus aculeatus	I		X	
	Gambusia affinis	NA		X	
	Rutilus rutilus	I	X	X	
	Pomatoschistus microps	I			X
	Mugil cephalus	I	X	X	X
	Liza ramada	I	X	X	X
	Lepomis gibbosus	NA	X	X	X
	Ictalurus melas	NA	X	X	
	Pseudorasbora parva	As	X	X	X
	Scardinius erythrophtalmus	I	X	X	X
	Stizostedion lucioperca	Ec	X		
	Silurus glanis	Ec	X		
	Tinca tinca	I		X	
Cray-fish	*Orconectes limosus*	NA	X	X	X
	Procambarus clarkii	NA	X	X	X

Carp dominated catches in the main canals (40% of the biomass). This fish has for long been the most widespread cyprinid in the Rhône river Delta, in water bodies with salinity less than 10 ppt (Crivelli 1981). For all species recorded in the canals, many age classes were poorly represented, especially among breeding adults, which suggests either age specific mortality, or more probably low reproductive success *in situ*.

In flooded fields, the most abundant species was the grey mullet *Liza ramada* (50 to 83%), whereas carp and goldfish were dominant in terms of biomass (63 to 97%). Specific richness and abundance of fish were higher in the fields closest to the spillways. Community structure in the flooded fields was comparable with that observed in seasonally-flooded marshes in the Camargue (Poizat and Crivelli 1997).

At the site scale recruitment was ensured either by breeding *in situ* or by entry with saltwater, thus providing prey for fish-eating birds.

Birds

The main objective for the birds was to provide suitable feeding areas for herons. A colony of tree-nesting herons became established at the start of the project on the internal dike system and is currently the main ornithological interest of the site. In 1996, 17 pairs of night herons bred successfully with an average of 2.2 young per brood (Figure 3). The colony consisted of 1560 nests of 4 species in 1999. The site now harbors one of the largest mixed colonies in France, including nearly a third of all the pairs of squacco heron breeding in Southern France, and the breeding success (mean number of fledged young per pair) is similar to that of the Camargue (Bennetts et al. 2000, Hafner et al. 2001). These data reflect the good quality of the site for the colony in terms of security (protection against predation, and the north-south orientation that protects against the prevailing winds and storms), nest construction (trees and bushes providing nesting sites and materials for nest construction) and the quality of the foraging sites within the colony home range (at least 800 ha of freshwater marshes within a radius of 5 km (Hafner and Fasola 1992).

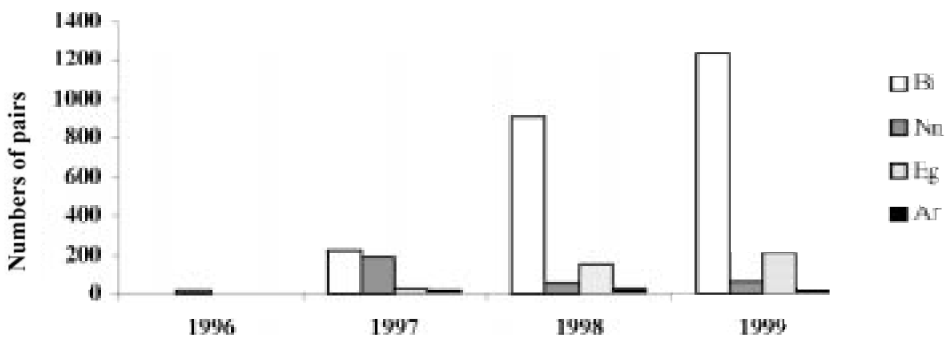

Figure 3. Species composition of the colony of tree-nesting herons in the Vistre Marshes. Bi: *Bulbucus ibis*, Nn: *Nycticorax nycticorax*, Eg: *Egretta garzetta*, Ar: *Ardeola ralloides*.

The numbers of herons feeding on the marsh varied greatly both temporally and spatially. Numbers were lower in 1999 (mean of 8 birds/visit) than in 1997 (82 birds/visit) and 1998 (63 birds/visit) probably because of the early desiccation of the system and the high salinity. The fish and crustaceans in the marsh were of a suitable prey size for the target species (herons) (Mauchamp et al. 2001 in press). The prey capture success rate estimated for the little egret from the number of pecks per minute was similar to values measured on other foraging sites (Lombardini 1999).

During each autumn the numbers of wintering and passage birds peaked and then declined profoundly (Figure 4) before increasing again at the end of winter. Two factors can explain this pattern. First, the rise in water level made the marshes unsuitable for herons and waders and second, the start of the hunting season in October was a major disturbance on this small site. Ducks and coot were much less abundant in 1999 than in the other two years (Figure 4). Wintering birds, however, were scarce throughout the Camargue in the winter of 1999.

Figure 4. Numbers of wintering and passage birds in the Vistre Marshes between 1997 and 1999.

In principle, the structure and deep water of the marsh provide a suitable day roost for ducks and coots. However, the unusual seasonal pattern of numbers for a wintering site suggest that the effect of disturbance caused by hunting had an overwhelmingly deleterious effect.

Conclusions

The lowering of the dikes in 1995 allowed the ecosystem to return to a successional path not requiring any further management works excepting the maintenance of the spillways. The restored marshes fulfilled the objectives of flood abatement and those concerning biodiversity and habitats. There were three main causes that can explain the deviation from the expected hydrological regime: 1) interventions by users who did not accept the constraints established by the project, 2) defective construction and maintenance of the spillways, and 3) insufficient data before the start of the project. To a great extent, solutions were found to overcome these difficulties.

Changes in the vegetation were in accordance with the initial hypotheses, taking into account the hydrology. They were controlled mainly by the selective pressure exerted by the physical environment on the species already present in the former polders. This was facilitated by the fact that many aquatic species already existed as a result of rice farming and had survived in the canals. Submerged macrophytes tolerant of salt and nutrient rich waters dominated. Water depth and poor water quality can explain only partly the decline in reedbeds. The high density of coypu probably added a further disturbance that was not taken into account in the initial project.

The objectives for the fauna were fulfilled with the establishment of the colony of tree-nesting herons and foraging areas similar to those in the Camargue. The potential value of the site as a refuge for wintering birds was confirmed, but hunting activity prevented this from being a reality from October to February.

The objectives of using the marshes for grazing and reed harvesting were not fulfilled for reasons of vegetation structure. These difficulties, added to the low quality of the marshes for hunting, made the project difficult to be accepted socially, and this was not sufficiently taken into account in the initial project.

This project demonstrated the feasibility of restoring wetlands based on spontaneous successional changes in ecosystems and minimal management following initial manipulation. To increase the probability of success and the predictability of expected changes, this approach requires considerable investment during the preliminary phases, which was far too limited in this case. The results must not be assessed over too short a time period and very precise "species" objectives must not be set. The wetland that was restored reconstituted the great natural fluctuations that occur under the influence of hydro-climatic conditions, and favored greater ecosystem resilience. Beyond inter-annual fluctuation the restored wetland will probably continue to develop toward the initial objectives, i.e. the establishment of a reedbed. These dynamics should on a long-term basis lead to the restoration of the organic layer of the soil. However the results of the restoration projects can be modified by changes in the management of the river and its floodplain. In riverine wetlands restoration and management should not be considered in isolation but be addressed on a relevant, wider, geographical scale.

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WETLAND RESTORATION IN RESPONSE TO PUBLIC NEEDS AND EXPECTATIONS: LAKE MAVROUDA

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Introduction

The multiple importance of wetlands for rural communities of developing countries has often been ignored, and drainage of wetlands has been tantamount to loosing their lifeline (Silvius et al. 2000). During the 20th century, large-scale conversion of wetlands to expand cropland and to control malaria was a common strategy in the Mediterranean basin. In several cases, this practice disenfranchised communities depending on wetland resources and stymied their economic development potential (Zalidis et al. 1999).

Wetland restoration plans must take into account hydrological, ecological and human interrelationships and needs, in addition to land-use planning, water management, economic development and conservation (Mitsch and Gosselink 1993). To ensure sustainable restoration and use of wetlands, it is essential to recognize the importance of cultural heritage, local practices and needs (Crisman et al. 1996). Thus, not only should local people be an integral component of the planning process and involved from the outset in its development (Anonymous 1996), but it must be recognized that public perception and driving forces can change through time (Turner et al. 2000).

The example of Lake Mavrouda in Central Macedonia, Greece, illustrates the evolution of public perception regarding the management of wetlands. The attitude of the local people regarding the wetland shifted three times during the last 50 years. During the 1950's and 1960's, the wetland was regarded as a wasteland that should be converted into agricultural land. The following decades saw a shift in public opinion towards restoration of the wetland, initially to support agricultural production (1970's) and finally in the direction of environmental conservation and associated economic activities (1980's). Such alterations in the public's perception were the result of continuously changing environmental and socioeconomic conditions in a broader context and the perceived needs of the people reflecting their knowledge, environmental awareness, and general policy shifts in competent government agencies. An important point in the case of Mavrouda is that the current restoration plan for the wetland not only has broad general acceptance, but also the local society is actively promoting the implementation of the project.

The objectives of this chapter are both to point out the role of the local community in the restoration of the Mavrouda wetland and to demonstrate utilization of the functional approach for sound long-term management of the wetland that accounts for environmental and socioeconomic constraints within the watershed.

The Watershed of Lake Mavrouda

The drained Lake Mavrouda lies on a plateau encircled by the mountains Kerdyllia, Volvi and Vertiskos, in the Prefecture of Thessaloniki, and is an example of human impact causing loss of wetland functions. Mavrouda's watershed (Figure 1) is almost circular and covers an area of 6,950 ha (Tsakoumis 1977). Formerly, a salt lake was located in the center of this basin and had a maximum water depth of 2.5 m. The mean altitude of the lake surface was 346 m, and its surface area was approximately 450 ha. The village of Mavrouda, with a population of 680 in 1961, was located close to the northeast shore of the lake (23° 28′ 01′′ E, 40° 48′ 08′′ N).

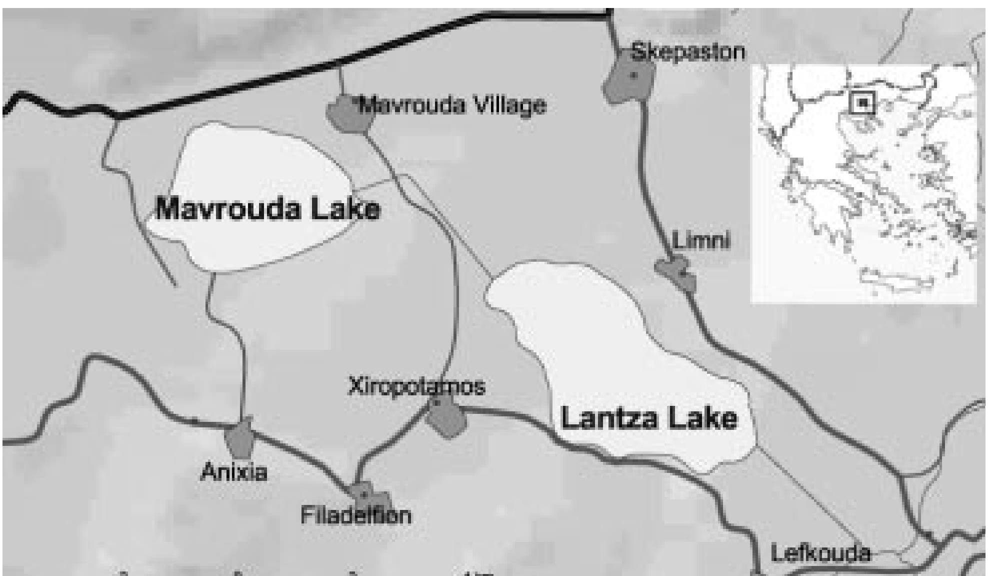

Figure 1. Lake Mavrouda before drainage in 1960.

Under pressure from the village since the 1920's, local authorities began to develop a drainage plan for the lake, and finally in 1960 Mavrouda was drained via a constructed network of ditches. It was expected that this action would increase the agricultural area.

Evolution of public perception regarding wetland restoration

Economic benefits from drainage were much smaller than expected because a large part of the lake bottom soils sustained small yields due to their salinity. Moreover, biodiversity and landscape diversity was degraded. Therefore, in the 1970's local farmers started to realize that perhaps their profession could best be served by a restored wetland which would provide some irrigation water and improve the hydrological conditions in the wider area. In other words, the first driving force for restoration was agriculture.

In the 1980's, however, a second driving force for restoration started to develop, namely that concerning the environment. This force was generated by the growing environmental awareness among wider segments of Greek society, including farmers. This awareness was supported by the changing policies of the European Union towards a reconciliation of demands for agricultural products and environmental goods and services. The rural population became increasingly aware that wetlands performed multiple functions which offered opportunities not only to optimize crop production, but also to develop other economic activities such as agrotourism and, in general, to improve life quality (e.g. landscape improvement, improvement of topoclimate).

The development of visitor-oriented nature conservation sites should help raise the profile of the region, as well as stimulate the local economy. In other words, wetland restoration should take into account the considerable potential for ecological and heritage tourism that could generate local income and employment benefits (Crisman 1999).

In spring 1993, the Mount Kerdyllia Development Association (a private non-profit organization with members from the greater Mavrouda area) requested scientific and technical assistance from EKBY to investigate the feasibility of restoring all or part of the former lake. EKBY's study concluded that, under current environmental and socioeconomic conditions, part of the wetland could be restored to provide the benefits and services desired by the local community (Chatzigiannakis and Zalidis 1993).

The functional approach for Mavrouda restoration

The development of a restoration plan for the Mavrouda wetland was a challenging task because of the environmental and socioeconomic constraints that, were identified in the watershed. For that reason, EKBY elaborated a study (Zalidis et al. 1998) where four restoration solutions were examined, based on an assessment of wetland functions. Two of these solutions (solutions I and II) examined the recreation of a wetland ecosystem to cover the area of the pre-existing one in its maximum and minimum coverage, while the rest (solutions III and IV) focused on the creation of a wetland covering less area but having greater water depths, taking into account the existing availability of water resources.

It was considered that the wetland should primarily be able to store a sufficient quantity of good quality water and should also support diverse food webs (through the appropriate support of primary and secondary production). Thus, the solution selected as most appropriate, was the one proposing the creation of a smaller wetland ecosystem, and the elaboration of excavations for the creation of both deep and shallow water habitats (solution III, Figure 2).

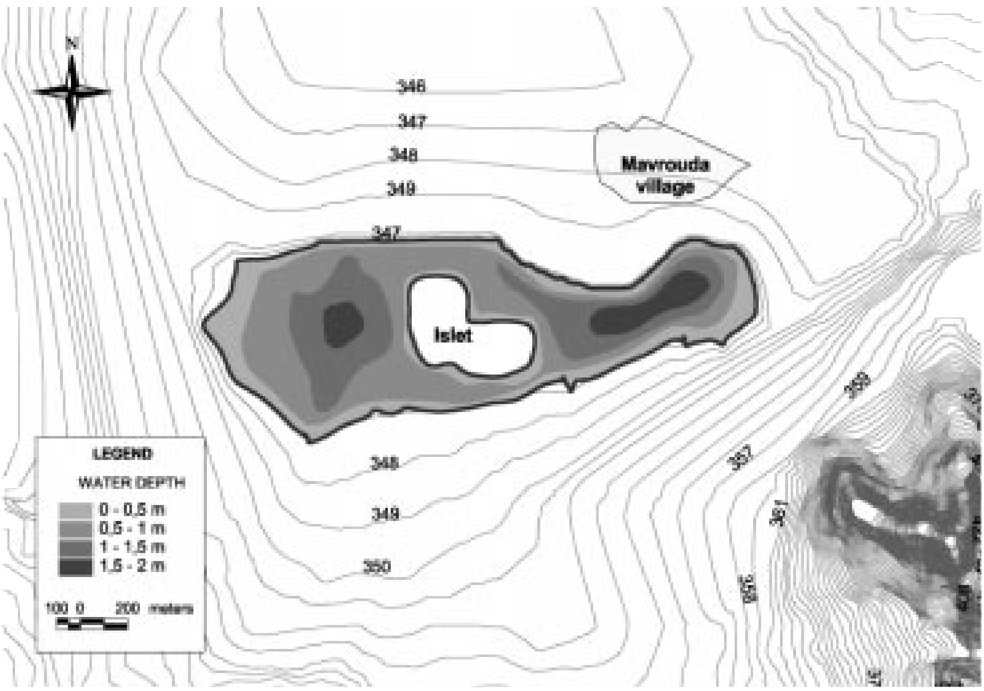

Figure 2. Mavrouda wetland following restoration according to restoration solution III.

The first step of the abovementioned approach was to set a reference condition to determine which functions and to what degree, should be restored, in order to establish a sustainable wetland ecosystem that could meet the needs and expectations of the local community.

The second step was to construct a Digital Elevation Model (DEM) for each proposed solution and to calculate a water budget in order to predict the structure and hydroperiod of the wetland after restoration. The characteristics of the wetland predicted for each restoration solution are presented in Tables 1 and 2.

Table 1. Predicted wetland characteristics for each restoration solution.

Wetland characteristics	**Solution I**	**Solution II**	**Solution III***	**Solution IV***
Total wetland area (ha)	382	267	105	126
Maximum water extent (ha)	382	267	90	113
Maximum water depth (m)	1.9	1.25	3.5	3.5
Maximum water volume (m^3)	3.8×10^6	1.6×10^6	930×10^3	1.4×10^3
Watershed area (ha)	6,800	6,800	3,730	3,730
Shoreline (km)	7.9	7.1	6.8	7.6

* Solutions III and IV include excavation in parts of the wetland and creation of an islet.

Table 2. Percentage of wetland area for water depth increments at high water for restoration solutions I, II, III, and IV.

	Solution I		**Solution II**		**Solution III**		**Solution IV**	
Water depth (cm)	**Wetland area (ha)**	**% of total wetland area**	**Wetland area (ha)**	**% of total wetland area**	**Wetland area (ha)**	**% of total wetland area**	**Wetland area (ha)**	**% of total wetland area**
0-30	30	8	30	11	17	19	18	16
30-60	97	25	97	36	18	20	21	19
60-200	255	67	140	53	32	37	47	41
> 200	0	0	0	0	22	24	28	24
Total	382	100	267	100	90	100	115	100

The third step was to compare the different restoration solutions and select the most appropriate one. To achieve this, a functional evaluation of each restoration solution was made (Table 3) using the Wetland Evaluation Technique of Adamus et al. (1987) modified to meet the needs of this study. The wetland hydrological and geomorphological characteristics delineated in the second step for each restoration solution were used as a basis for the functional evaluation of the proposed solutions.

Table 3. Degree of restoration of wetland functions, expected to be achieved by each solution.

Wetland Functions	Solution I	Solution II	Solution III	Solution IV
Modification of flood phenomena	Low	Low	High	High
Aquifer recharge	Low	Low	Low	Low
Nutrient removal/transformation	Moderate	Moderate	Moderate	Moderate
Sediment and toxicant trapping	Moderate	Moderate	Moderate	Moderate
Food web support	Low	Moderate	High	High
Net primary productivity	Low	High	High	High
Secondary productivity	Low	Moderate	High	High

Functional analysis, when applied to lost wetlands, is an effective tool for revealing past errors in the management of water and soil resources and to prevent future errors when designing wetland restoration schemes based on the sustainability of natural functions (Zalidis and Gerakis 1999). In the case of Mavrouda, solution III (Figure 3) was deemed the most appropriate for the sustainable restoration of the ecosystem since wetland functions could be restored to a desirable degree and the expected water balance will maintain appropriate water salinity in the wetland (Zalidis et al. 1997, Zalidis et al. 1998).

The social approach to Mavrouda restoration

Wetland restoration must result in socially and culturally desired ecological characteristics within a local context. In other words, the final outcome of restoration must include the various stakeholder interests in order to have meaning to society. Unfortunately, conflicts of interest of stakeholders in a watershed usually result in wide disagreement about the desired physical appearance of the wetland, the functions performed and the goods and services produced (Wyant et al. 1995). However, stakeholders can develop cooperative relationships as a means of achieving resource management goals (Berkes et al. 1989). If one measure of sustainability is the establishment of responsible stewardship over the natural resources, then such cooperative arrangements are essential (Lejano and Davos 1999).

Mavrouda is one of the few cases in the Mediterranean basin where restoration of a wetland was demanded by the local community. Thus, the implementation of the project had general acceptance by stakeholders in the area from the beginning, because the restoration objectives resulted from discussions with the local community. The scientifically best restoration solution was communicated to the local community and accepted widely. Weighing alternatives and ultimately choosing among trade-offs is fundamental in realizing sustainable restoration projects in a pluralistic society. Thus, the concern and active involvement of the local society in implementing the restoration project resulted in considerable pressure being placed on decision and policy makers of the region, and today the restoration project is ongoing.

Collective action, as in the case of Mavrouda, is important for wetland restoration, both because of involved interactions among land owners/stakeholders and because of the cost saving and enhanced environmental benefit that can be achieved at a larger scale (Hodge and McNally 2000).

Conclusions

From ancient times, the needs and expectations of society have dictated the fate of natural resources, including wetlands. The Mavrouda wetland is a classic case where local community needs resulted in the drainage of the wetland in the 1960's, then promoted restoration of the ecosystem a few decades later. It is more than obvious that the ecological characteristics of the restored wetland should reflect the environmental and socioeconomic status of the watershed. The functional approach used in the restoration planning of the wetland took into account not only the environmental constraints of the area, but also the vision of the local community for the expected project outcome. Thus, the restored ecosystem will be able to meet the needs of the local society in terms of environmental stewardship and socioeconomic development as they evolve together in the future.

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RESTORING A COASTAL WETLAND: L' ALBUFERA OF VALENCIA IN SPAIN

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Introduction

L' Albufera pond is an internationally important coastal wetland, eligible for inclusion in the Natura 2000 network, and covers about 23 km^2, approximately 12 km from the city of Valencia. It consists of three ecological units, namely, a coastal sand-dune barrier, an open water area and a marsh. In addition to its ecological values, it is also important for its fishing, aesthetic, agricultural, cultural and recreational values. Unwise use of the resources of the wetland and its basin since 1960 has caused serious degradation of several of the wetland's structural and functional features, including water quality, sediment deposition, dune morphology, coastline stability and biodiversity. Growing environmental awareness of the local people motivated a series of restoration projects in the last decade. These projects were carried out via two strategies: The first aimed at implementing passive management measures such as regulating land uses, and limiting the inflow of pollutants. The second strategy focussed on active means of restoration such as revegetation with autochthonous species from a specially created nursery, sediment control measures and dune stabilization measures. The overall goal of these strategies was to lead the wetland and dune ecosystems into a process of self recovery. This was achieved to a large extent. However, it is unknown whether full recovery will occur without control of the effects of some negative factors.

The term *ecological restoration* has been widely used to define all kinds of management options to re-establish or accelerate the development of communities in degraded or otherwise altered ecosystems, and to reverse or ameliorate various forms of environmental damage. Ecological restoration has deep roots in more generally conceived forms of environmental management such as forestry, game management and certain forms of agriculture and landscape design. Current restoration, however, is a comparatively recent development, which has reflected the application of the principles of the ecology to traditional forms of rehabilitative environmental management. In some cases, rehabilitative management had ecological and historical goals as objectives rather than utilitarian or aesthetic ones, and there was emphasis on creating a system that resembles the original system as closely as possible. Some of the earliest projects in the 1920's and 1930's reflected this concern for authenticity and re-created examples of native ecosystems (Dobson et al. 1997, Montalvo et al. 1997, Palmer et al. 1997).

Nevertheless, it is worthwhile to clarify some of the terminology in order to define concepts more exactly. *Restoration* of ecosystems means, exactly, that we intend to return the damaged ecosystem to approximately the same structural and functional condition it had before impact (Parker 1997). Since each ecosystem is the result of a sequence of climatic and biological interactions that are unlikely to be repeated in precisely the same way, exact duplication of the condition before disturbance will be rare. In this strict sense, it is clear that, although restoration is rare, it can play a significant role in the conservation of natural ecosystems in all parts of the world. More often what we normally obtain in most of the so-called restoration projects is re-establishment of some ecological conditions that resemble but do not duplicate the pre-damage ones. In these cases, it is more correct to refer to them as "rehabilitated", "recovered" or re-established ecosystems, a term that translates into Spanish as "recuperación de ecosistemas" instead of "restauración de ecosistemas".

Thus, we do not recommend the term re-creation because it may lead to confusion.

General features of the L' Albufera Coastal Pond

L' Albufera is both the most important wetland of the Region of Valencia and an important coastal pond in Spain. It is 12 km south of Valencia, and presently has a surface area of 23.2 ± 0.1 km^2. It can be divided into three ecological units:

- *Coastal barrier* (La Devesa): A wide strip of sand some 28 km long, beginning at the Turia river mouth and reaching the Mountain of Cullera. It presents an important dune system forming two alignments between which there is a long interdunal depression. This depression consists of several small endorreic basins known as "Malladas". These are characteristically marshy since they flood sporadically; salt concentration increases when the water evaporates.

  The inner (landward) dune alignment is 300-600 m wide. It is the oldest and is stabilized by vegetation (pines and shrubs). It is a product of the easterly palaeowinds. The outer seaward dune, next to the beach, is 150-300 m wide and is formed by ever moving transversal dunes.

- *Open Wate*r: It covers 2,837 ha and has a depth of 0.5-2.0 m. Its maximum width is 8 km. It is a freshwater pond with a pH of 9-10. The water comes mainly from surface sources, residual irrigation water and pot holes springs (ullals) fed by fresh groundwaters.

  There is an abundant, dense biomass of algae and aquatic macrophytes (of the genus *Potamogeton*, *Chara*, *Myriophyllum*, *Ceratophyllum*).

- *Marsh* (Marjal): it is a paludal area of 22,300 ha. In the past, it was much larger. Its substrate is rich in semi-decomposed organic matter of both natural and anthropogenic sources. The natural marsh is common along the shores of the pond and in the islands ("Matas") that appear inside the pond. The latter are the last vestige of the natural marshes that occupied large areas and are essential for the conservation of biodiversity, since they are the habitat for unique plant species providing nests, shelter, and food for several species. There is a vegetation complex that includes floating species such as *Lemna gibba*, rooted aquatic species such as *Nymphaea alba*, *Myriophyllum* sp. etc. These, together with free floating species of the Potametea and Ceratophylletea class and reedbeds form the emerged plant community of the islands.

L' Albufera Pond drains through the inlets (golas) of Perelló, Perellonet and Pujol, all of which are constructed (Sanchis 1998).

Habitats of great interest have evolved in these units, which have favored development of a diverse flora (more than 800 species, many catalogued as rare, endemic or threatened) and fauna. Waterfowl are notable among these groups, since 250 species use the area at some time during their life cycle, and of these, 90 nest here regularly. Many species are listed as rare or threatened; 40 are included in Annex I of the European Directive on Birds, and the number rises to 80 if we include those of the National Catalogue of Species Endangered or Threatened by Extinction.

L' Albufera is an invaluable addition to the local landscape by being located in an semi-arid zone and very close to urban centers. It is also a wetland where several traditional resource uses are still practiced. Fishing, agriculture (rice growing) and game utilization are important for the local economy.

All these ecological, economic and cultural values have motivated the regional government to provide L' Albufera with legal protection since 1986 under the status of a Natural Park and with its inclusion in the Catalogue of Wetlands of the Region of Valencia in the year 2000. These values have also earned it international recognition since it is included in the list of "Internationally Important Wetlands" according to the Ramsar Convention. It has also been declared a "Special Protection Zone for Birds" (Directive 79/409 of the European Union on Conservation of Birds) and is part of the Natura 2000 Network (Directive 92/43 of the European Union on the Conservation of Natural Habitats and of Wild Fauna and Flora).

Main environmental impacts

From the human point of view, besides the intensive agricultural activity carried out around most of the wetland, the densest urban-industrial belt of the Region of Valencia lies around the pond. Nearly 1.5 million people live in its immediate surroundings and more than 5,000 industrial sites are found in the basin. This, together with intense tourism/-residential development in several sectors of the coastal zone, provides a nearly continuous urban belt around the wetland.

This enormous anthropogenic pressure has generated multiple degradation factors that have negatively affected the natural values of the area in the past few decades. Conflict has developed between the growing social will to conserve this natural area and the need to maintain human activities. In the last few years, specific legal instruments destined to protect and assess the territorial uses, and to invest public funds in environmental restoration and infrastructures and utilities have been developed. It is hoped that these instruments will help recover the traditional balance between man and environment that was characteristic of L' Albufera up to the first half of the 20th century.

The resource that has suffered the greatest impacts has been the water itself. During the first half of the 20th century there was a net loss of aquatic aerial extent due to infilling for land reclamation for agriculture (*Aterraments*). In the 1970's, other ecological units such as the coastal barrier (Devesa) were altered by construction unrelated to agriculture. Thus, the mobile dune alignement was destroyed (in Southern Europe this has been a normal practice, and since the 1960's, 75% of coastal dunes have disappeared). Also, some infrastructure (buildings, roads, parking lots) was built. Part of the "malladas" were filled to level the terrain, and eucalyptus trees were planted to help drain it. Uncontrolled recreation activities also caused degradation in the area. The beach has also suffered erosion, associated with facilities of the commercial Port of Valencia located upstream. The water has been polluted with domestic and industrial wastewaters (coming from the periphery of the system).

Until the beginning of the 1960's, the pond displayed high water quality, although the system had already undergone some alteration. It supported a rich biota. In the 1950's, the construction of sewage network systems in neighboring towns and the enormous industrial boom in the metropolitan area of Valencia and in other parts of the drainage basin brought great quantities of urban and industrial wastewaters into the pond. This pollution increased steadily until the 1980's, when a serious pollution crisis was declared. Pollution accelerated eutrophication. Phytoplankton biomass (comprised almost exclusively of Cyanobacteria) increased tremendously (Vicente 1992) and restricted light penetration to a few cm below the water surface. The increase of non-point agricultural pollution did much to aggravate the situation: submerged macrophytes started to disappear and so did other aquatic biota. Biodiversity was therefore decreased. In this manner, most of the invertebrates associated with aquatic vegetation disappeared, and with them, faunal elements higher in the foodweb. Species richness was reduced to a minimum, especially for fish and waterfowl.

The disappearance of the grass fields was accompanied by the loss of their protective role; sedimentary and hydrodynamic regimes of the pond were radically modified. From a situation of positive sedimentation around the islands (favoring their expansion) a negative status has developed in which there is a net loss of island surface area.The island surface has been reduced by 20% in the last 30 years (Argiles 1997) due to erosive wave effects on the shores.

Corrective measures and restoration programs

To address the deterioration of the wetland, various actions have been carried out by local and regional administrations, often partially funded by the national administration. Most funds, however, have been obtained from European Community Programs (FEDER, LIFE etc.).

It is important to note that there has never been an integrated restoration plan for Albufera. All actions have been very specific and had to be implemented urgently. Larger and more costly goals (improvement of the quality of the water in the pond, for example) were originally guided by sanitary rather than by ecological reasons. In part, they were also mandates from the European Directives (Water Purification etc.) rather than from an awareness of the need to conserve these ecosystems.

There have been two strategies for rehabilitating the ecosystem: the first is spontaneous or passive regeneration based primarily on developing of legal instruments to control the activities (prohibitions, use limitations, access restrictions) in order to achieve conditions of minimum disturbance over the long term and thus be able to recover some habitats. This is the case of the dune zone and water quality. In the latter, technological and economic intervention in the activities having impact has been decisive. In this way, the problem has been approached from its origins, that is, trying to eliminate pollution point sources and allowing self-cleansing of the system.

The second strategy has been active and clearly interventionist to achieve more rapid and controlled results. This has been applied in the most affected places, where it was not only necessary to restore morphology, but also dynamic processes. These dynamic processes are the basis for maintaining the functioning of the system and where it was very difficult to eliminate the cause of degradation. This has been the case of the shoreline front where restoration of the morphology and vegetation has been relatively simple. However, restoration of the functions has not always been successful.

Dune regeneration

The dune system was affected by urbanization initiated in the 1960's (coinciding with the Spanish tourist boom) and stopped at the beginning of the 1980's thanks to popular initiative, although the ecosystem was already seriously altered.

In this manner, part of the public land that had been altered passed to private owners. The first dune alignment had been destroyed, and urbanized areas proliferated (apartment buildings, facilities, beach-side promenade, parking lots), and there was an enormous public debt due to poor economic management.

Moreover, the public use of the Devesa as a traditional green zone for the city of Valencia and its surroundings developed. Its access was facilitated by the Valencia-El Saler motorway. The road network and parking lots, together with the scarcity of urban and metropolitan green zones, attracted a great number of visitors. This mean more fires, soil erosion and the impossibility of regenerating the natural ecosystem.

As previously mentioned, the local government initiated a recovery of this natural area. In 1979, the "Urbanization Plan" was voided, the land was reclassified as "special protection area" and the following year an entity especially oriented towards the management and conservation of this zone (Oficina Técnica Devesa-Albufera-OTDA) was created. An environmental diagnosis was made and a plan drafted to guide future actions. This plan was reviewed 13 years later (1996), to evaluate the degree of compliance of the plan and the aims achieved (Andreu and Viñals 1996). It is a positive initiative, since it is not common to review and evaluate the efficiency of such plans.

The administrative management of the OTDA was centered on stopping urbanization, territorial zoning of the Devesa, stopping indiscriminate access, and trying to regulate possible uses and visitors spatially according to the carrying capacity of the different ecosystems. At the same time, awareness campaigns were carried out to involve the population and demonstrate to them the social benefits to be gained from the proposed radical change of land use.

The most important restoration action undertaken by OTDA was the natural regeneration of mobile dunes and "malladas". To achieve this, municipal nurseries for autochthonous plants were created, and plantings were carried out in the ecosystem.

The goal for mobile and fixed dunes was to recover as far as possible a pristine morphology, at the same time decreasing and stopping the progressive degradation of the brush and pinewoods. To achieve this aim, negative infrastructures (hydroelectric buildings, parking lots, roads and ducts) were eliminated along with exotic species (*Carpobrotus* and *Agave*). Access for both pedestrians and cars was reduced and dune morphology was reconstructed through the introduction of plants from the autochthonous plant nursery. Later, recovery of faunal populations and maintenance of vigilance and control of the area was carried out.

Dune regeneration has had spectacular success, and its methodology has been and is being used and recognized by Spanish and foreign experts.

It is thought that with a significant economic investment, it should be possible to recover the first dune front completely.

The preventive and conservation measures (fire prevention, limited access for pedestrians, reduction of paved traffic areas) are considered correct and should even be extended.

Recovery of the morphology of the "malladas"

The most frequent problem of the "malladas" has been infilling of the depressions with at least 0.5 m of sand leading to basin shallowing. In addition, several malladas were divided longitudinally by a central elevated road. Two of them ("La Rambla" and "Raco de l' Olla") have suffered even greater alterations. The first had been used as a rubble dump, and the second one was occupied by a horse race track, which meant it had completely lost its morphohydrodynamic features.

Recovery began by eliminating exotic species (in this case, *Eucalyptus* sp.), extracting the sandy infill, and eliminating existing asphalt (roads).

All of these actions were carried out in the Racó de l'Olla. In this manner, with joint actions of local and regional governments, work was initiated to eliminate the race track infrastructures and the infills. The depression was re-excavated, and two pools were designed. At the same time an unaltered saltmarsh was preserved.

Finally, the area was divided into two parts: an Ornithological Reserve and the Information Center of the Natural Park with a pool that can be visited.

The success of the waterfowl reserve has been spectacular. Bird populations increased markedly and many species have reappeared and reproduced since 1992.

Improvement of water quality

In 1994, all administrations (including the national) undertook the Integral Restoration of the Albufera Plan. The basic aim of the plan was to design and build the necessary sanitary and purification infrastructure to allow for the improvement of water quality in the pond. A perimeter sewage collector (West Collector) was constructed to carry domestic and industrial wastewaters of all towns surrounding the pond to a residual water treatment plant in Pinedo (north of Albufera).

This plan also included the construction of other water treatment systems in the drainage basin of Albufera (Quart, Torrent, Albufera Sud, Algmesi-Albalat, Sueca, Cullera) and in other locations along the shore (El Saler, El Palmar, El Perelló and El Perellonet).

At present, part of the planned systems have been constructed and have reduced the volume of wastewaters reaching the pond. This has had a positive effect on water quality. An increase has been marked in the diversity of zooplankton, couple with a decrease in phytoplankton biomass (a common index of trophic state in aquatic systems) and partial recovery of water transparency. The improvement is such that in the last few years, during February and March when conditions are most favorable for phytoplankton, there have been episodes of "clear phases" lasting 3-4 weeks, during which the water is so clear that the bottom can be seen. These episodes coincide with an explosion of zooplankton (especially *Daphnia magna*).

There are still some water treatment plants to be built, especially in the southern zone (Albufera-Sud, Algemesí-Albalat, Sueca y Cullera) and the secondary branches and sanitary networks along sectors of the West Collector. The level of purification (especially the decrease in N and P concentrations) must be increased in the outflows of most of the current systems, according to specifications set by the pertinent Directives of the European Union.

The physico-chemical and ecological parameters of the water are regularly monitored. Determinations of temperature, oxygen concentration, pH, conductivity, turbidity, BOD, concentrations and composition of phytoplankton and zooplankton, chlorophyll, nutrients, presence of pesticides, heavy metals and other toxic substance are part of the monitoring program.

Rehabilitation of the littoral front

The rehabilitation of the shoreline area of the northern sector of the barrier (Pinedo zone) has been undertaken in co-operation with the national administration. In the dune zone, similar actions have been executed. Existing buildings on the coastline have been demolished, and recreational areas further inland (not on the first line) have been properly located. In this manner, the first line of dunes may be reconstructed.

Beaches have been the environments where the most drastic restoration activities have been carried out. These environments suffered a serious decrease in sediment deposition all the way to the central sector of the barrier. These were associated with both natural and anthropogenic actions. In the first place, the dams of the Turia river, the main supplier of sand to the study area, retained sediments for centuries. However, the most dramatic effects have resulted from infrastructures of the Port of Valencia and their successive extensions.The coastline has recovered and its slope has become steeper as a result of coastline erosion.

Actions carried out by the central government with FEDER funds have focused on re-establishing the morphology of the zone by artificially regenerating the beaches. However, it has not been possible to reinstate the marine morphodynamic processes, nor will it be possible, as long as the Port exists. For this reason, actions to restore the beaches will be, at best, short lived and may even have some unexpected impact on another part of the coast. In our case, it has been positive, since the sand eroded from the regenerated zones supplied sectors farther south lacking this material.

The OTDA carried sand from the northern zone of the Valencia Port (Malvarrosa Beach) to reconstruct dune relief mechanically (modeling with bulldozer).

Recovery of habitats and endangered endemic species

There are several endangered endemic species of fish in the Albufera. These have received special attention in order to prevent their eventual disappearance and to enable their reintroduction into their habitat. *Valentia hispanica* has recovered in aquaculture facilities made using funds obtained from a European LIFE project. Various reintroductions of these fish from 1991 to 1995, have been made in the "malladas", in the Racó de l'Olla Reserve and in various temporary ponds of the Natural Park.

Evaluation of actions

The principal actions have been focused on the following aspects:

- Conservation of abiotic and biotic elements of the least altered areas.
- Elimination of impacting infrastructures, restoration of the morphology and dynamic processes
- Re-vegetation of degraded or restored areas, landscape enhancement and improvement of water quality.

These interventions are positive, in general, especially considering the brief time elapsed and the scant budget allotted. Dune regeneration and improvement of water quality are notable positive aspects. Sometimes the goals have not been achieved, not due to a faulty technical procedure but rather to adverse natural events (forest fires, sea storms) that impeded progress of the work.

Other interventions have temporally alleviated the problem, but they need constant attention to be able to maintain not only biotic processes but also habitat morphology. This is especially true for beach regeneration, a problem linked to factors that lie outside the management scope of L' Albufera. Control of these factors would be necessary to avoid its erosion.

Conclusions

L' Albufera represents a model of the Mediterranean coastal wetland regarding the environmental consideration that it has deserved. As a conclusion to this study, the following could be highlighted:

- Before the 20th Century, its exploitation was harmonious, and was mainly based on traditional agricultural and fishing techniques.
- From the sixties onwards, it was strongly degraded. This lasted for more than 15 years, forcing the system into a deep ecological crisis.
- In the mid seventies, as was the case in the rest of the Mediterranean countries, specific environmental protection rules were applied, putting a stop to degradation. However, there was no social backing for this strategy as yet. Regional Administration was still applying extremely passive measures for its conservation.
- The last decade of the 20th Century saw a more active conservation behaviour. It was originally driven by sanitary issues (to improve water quality) rather than ecological purposes. However, this has evolved into recovery projects based exclusively on ecological objectives. This has happened fairly slowly and it has been caused by the increasing awareness of the population towards environmental issues. These projects have been mainly funded with the decissive help of the European Commission.

Finally, we shall add that results obtained after implementing the restoration projects have been technically optimal. The wetland is currently included in the "Internationally Important Wetlands" according to the Ramsar Convention and it has also been declared a "EU Special Protection Zone", under the Birds Directive. It is included in the proposed list of sites under the Habitats Directive, which will contribute to the creation of the Natura 2000 network. Besides, the public awareness of the importance of the wetland keeps increasing, and a collective consciousness has arisen among citizens, now holding this wetland as an irreplaceable natural heritage.

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RESTORATION OF TIDAL MARSHES IN GUADALQUIVIR RIVER ESTUARY, SW SPAIN

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Introduction

The Guadalquivir river is the largest river in Southern Spain with a watershed area of 63,972 km^2 and an average annual discharge of 7,230x10^6 m^3. The climate of the region is Mediterranean, with an average annual precipitation of 595 mm. About 80% of precipitation falls in the winter months. July and August are practically rainless. Precipitation patterns are irregular: some years falling below 300 mm and other years exceeding 1,000 mm. Intense spells of precipitation may cover large sections of the watershed, with values up to 60 mm/day or 200 mm/month resulting in high river discharges. Annual volumes range from 22,235x10^6 m^3 (1962-63) to a minimum of 981x10^6 m^3 (1948-49). Recurrence times for peak flooding in Seville, at the beginnings of the estuary and about 80 km towards the sea, give 4,800 m^3/sec every 10 years and 15,652 m^3/sec every 500 years (Ministerio de Medio Ambiente 2000).

Seville suffered repeated disasters due to river floods. There are records of 100 disasters over the last 10 centuries. The port was destroyed, vessels were grounded, buildings collapsed and the human population, besieged by waters for months, suffered badly from pestilence and other illnesses, and starvation. The well-documented 1892 flood, with an estimated discharge of 9,800 m^3/sec caused the water level to rise by 10 m.

Large river discharges flooded an area of about 100,000 ha in the lowlands from Seville to the sea, depositing a silt layer on the old marshes and agricultural land. The Roman geographer Avienus (4th century) described a large lagoon (*lacus ligustinus*) in the marsh area, what may have covered as much as 250,000 ha. The accretion rate of the silt deposits has ranged from 1 to 3 mm/year, depending on the site, for the last 2 millennia and the area was reduced to 136,000 ha by mid the 20th century (González Arteaga 1993).

The construction of irrigation schemes from 1920 to 1990 finally reduced the preserved surface to 23,000 ha of inner marshlands (Doñana Marsh) from the estuary and confined relict tidal marshes along the estuary. Although tides spread 108 km upstream, saline or brackish water rarely enters more than 40 km and tidal marshes are restricted to the lower 25 km and to the estuary banks.

Since 1980, the dredging of the Guadalquivir river navigation channel towards the Seville port has seriously affected 8 km out of the 15 km of tidal marshes on the left bank of the estuary. Dredged sediments (largely sand and silt) were pumped to rectangular pools built on the tidal marsh forming 1-3 m high deposits. The estimated volume of dredged sediment deposits was 4 million m^3.

As a result, the former marshes were transformed into silt plains with temporary pond formation during the rainy season or to mobile dunes (where sand sediments prevailed). Semi-natural vegetation developed, dominated by species adapted to sandy substrates or saline soils commonly found in the marsh or the banks of the estuary. Perennials were dominated by succulent Chenopodiaceae (*Salicornia*, *Sarcocornia*, *Atriplex*, *Salsola*, *Suaeda* and *Arthrocnemum* species) and *Halimione portulacoides* or *Limonium ferulaceum*. *Tamarix canariensis* plants also invaded the area forming small thickets. Small ditches were lined up with *Phragmites australis* and *Scirpus maritimus*.

Spartina densiflora, a South American neophyte was introduced to the South of the Iberian Peninsula early in 20th century (Tutin et al. 1980). It invaded brackish tidal mud flats from South Portugal to the Gibraltar Straits, causing significant changes in vegetation structure with respect to tide range and competing for space and resources with native *S. maritima* (Castillo et al. 2000). *S. densiflora* now forms a dense band 50-75 m wide on both banks of the Guadalquivir estuary. This species also penetrates along the dendritic channels, building dense stands in their upper borders.

The objective of this chapter is to briefly describe the alterations effected by human interventions in the Algaida Marshes and to present the efforts which were carried out to restore these tidal marshes.

The Algaida Marshes

The Algaida Marshes on the left bank of the Guadalquivir estuary (36° 54′ N; 6° 19′E) belong to the Doñana Natural Park (Figure 1). The estuary may be classed as mesotidal with semidiurnal tides and a maximum range of 3.30 m. However, the marked rise of water level in river floods results in complete inundation and silt deposition over the marshland area. Because of the suspended solids carried by the water, the river has slowly changed its course, leaving old channels lined up with levees and small spits. These morphological features may be identified in the 1940 and 1956 aerial photographs.

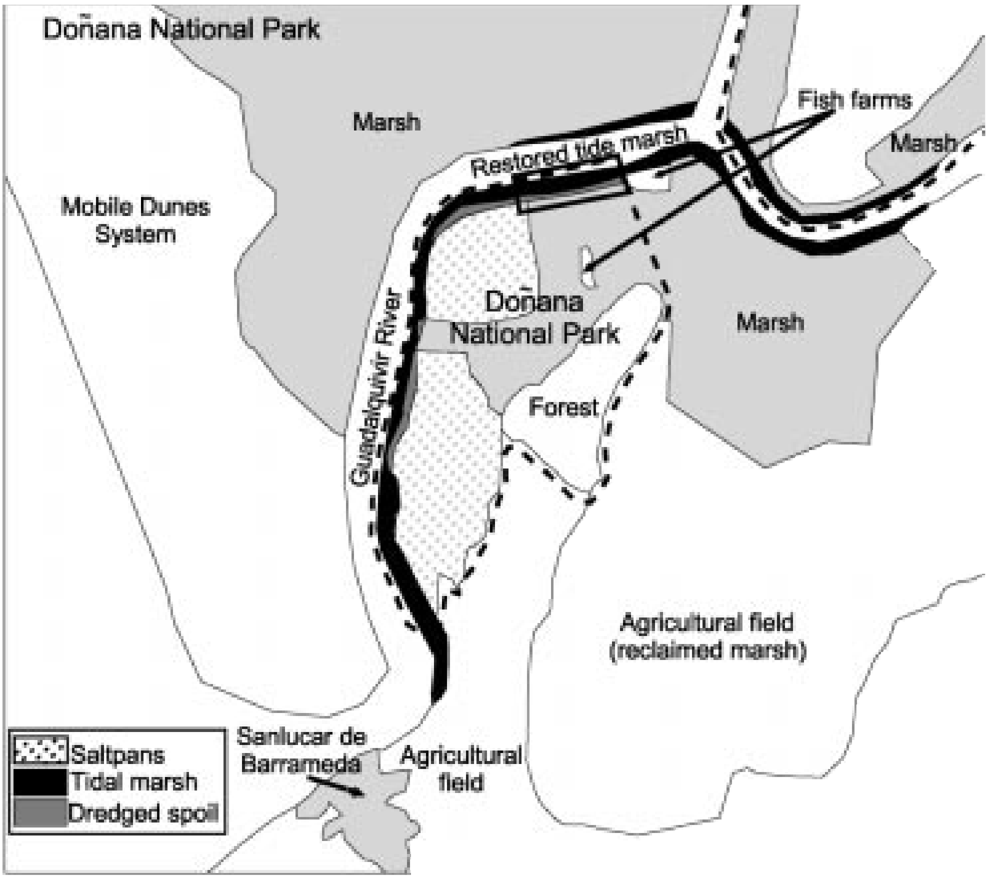


Figure 1. General layout of the restored tidal marsh and its surroundings.

Ancient interventions still recognizable in the air photographs, were the remnants of Roman salt pans, which are noticeable to the present day as shallow elongated ponds. Early in the 20th century, small salt pans were built in the area.

Between 1957 and 1960, a dike was built running parallel to the river, thus preventing brackish water from entering the Algaida Marsh in order to control soil salinity in the pastureland (Figure 2). In the 60's a wide channel equipped with weirs was dug to drain the marsh. This intervention was part of a State plan carried out in the 60's and 70's to turn 110,000 ha of Guadalquivir Marshes into cultivated land.

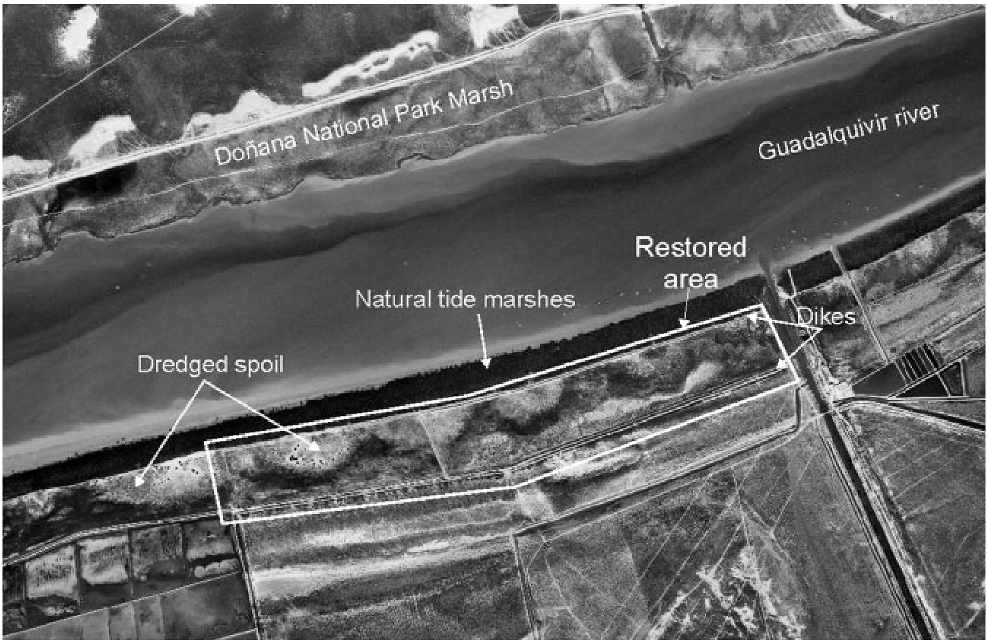


Figure 2. Air-photograph of the restored Algaida Marsh with its perimetric dikes

In the 70's, additional interventions were conducted in the Algaida Marsh: Ponds were constructed for aquaculture and ditches were dug to secure water supply from the estuary. In 1985 the dredging of the Guadalquivir river to make a deeper navigation channel began. An earthen dike was built, parallel to the existing inner dike constructed in the 1950's. Dredged sediments (largely sands) were deposited to heights of 0.5 to 2.8 m over a band of tidal marsh about 200 m wide and 2.5 km long, largely covered with *S. densiflora* and possessing abundant waterfowl and a nesting colony of spoonbills (Valverde 1960). The old marsh was turned into a "derelict land", losing much of its biological values.

Over the last 50 years, 60% of the original Marsh surface disappeared. A summary of the area changes per land cover/use type is shown in Table 1.

At 1993 the Spanish Ministry of the Environment initiated actions to mitigate or compensate wetland loss, in a way similar to other governments, following the lead of the 1992 Rio Conference (Hey and Philippi 1999).

Blueprints for restoration plans on 600 ha of tidal marsh were produced 1 year later (Ministerio de Medio Ambiente 1994), but private owners of the area opposed those plans. A legal decision sanctioned the exclusive State rights on the marsh sector adjoining the estuary, previously filled in with dredging material. At 1997 the Ministry pursued plans for ecological restoration of 52 ha of Algaida Marsh on a band about 250 m wide for 2 km along the estuary.

Table 1. Recent changes in Algaida Marshes (Ministerio de Medio Ambiente 1994).

Land unit	1956		1973		1992	
	ha	%	*ha*	%	*ha*	%
Tidal salt marshes	246	9.5	171.4	6.6	74.4	2.4
Marshes	2,207	85.2	1,463.7	56.5	906.5	35
Saltpan	137.1	5.3	465.4	18	864.7	33.4
Cropland			492.7	19	492.7	19
Fill of dredged sediments					139.7	5.4
Aquaculture					40.5	1.6

Specific goals for restoration

The first goal in the restoration of the Algaida Marshes has been to re-establish the ecological functions and the linkages between the biotic and abiotic components so as to achieve self-sustaining ecosystems. This is in tune with the proposals of the Society of Wetland Scientists (2000) for ecological restoration and goes well beyond former approaches to reclaim derelict lands (Oxenham 1966).

In fluctuating natural systems, such as marshes, restoration does not imply precise reconstruction of pre-existing communities or landforms. Rather, it implies process restoration, recovering prevailing ecological interactions if they were lost, letting self-organizing systems to rebuild morphology and species to enter the community. If the tidal regime was lost or flooding no longer occurs, the basic character of the area has changed and restoration faces a re-creation of a natural area (Broome and Craft 2000). The Algaida Marsh has been covered with dredging deposits. This caused a complete alteration of the character of the marsh. Therefore, the restoration actions may be considered as re-creation actions.

The second goal of the restoration was to promote local biodiversity at all levels. This was because Doñana Park is becoming an ecological refuge to endangered species at a regional level. The idea of restoring with an eye on menaced or endangered species can be traced to Bradshaw and Chadwick (1980). Doñana Park policies have focused on a few (most endangered) species, adapting habitat and resources to match their need through specific plans. In the Algaida Marsh it was decided that the best protection policy was to make available to organisms a mosaic of habitats as diverse as possible, by combining aquatic and terrestrial media and different substrates.

The restoration was accomplished in two steps:

- Removal of dredged material from the fill and restoration of tidal flow.
- Diversification of habitats and addition of soil.

The area was divided into two sectors on the basis of the intensity of restoration interventions required:

Sector 1: High intervention level, with an intense remodeling action to build the mosaic of diverse habitats in the north sector.

Sector 2: Low intervention level, just inducing geomorphological processes in the south sector.

Restoration plans were drawn by the Department of Plant Biology and Ecology of the University of Seville, under the responsibility of the authors, in 1998. The Director of the Park of Doñana approved the plans in 1999. In 1999 the Spanish Ministry of the Environment through the Demarcacion de Costas of the province of Cadiz assigned the works by dividing them into two projects: the fill removal and the ecological restoration, that were carried out during 1999-2000.

The restoration steps

Step One: Removal of dredged material and restoration of tidal flow

In September 1999 the fill, about 600,000 m^3, was removed from the marsh. It left a flat mud surface slightly sloping to the river (0.3% slope) at a level accessible to tidal flooding with maximum depth ranging from 0.66 to 1.26 m (Figure 3). Inner and outer dikes were preserved and rebuilt. Sixty percent of the fill material, largely sands, was ferried to neighboring agricultural land to improve soil quality for crops.

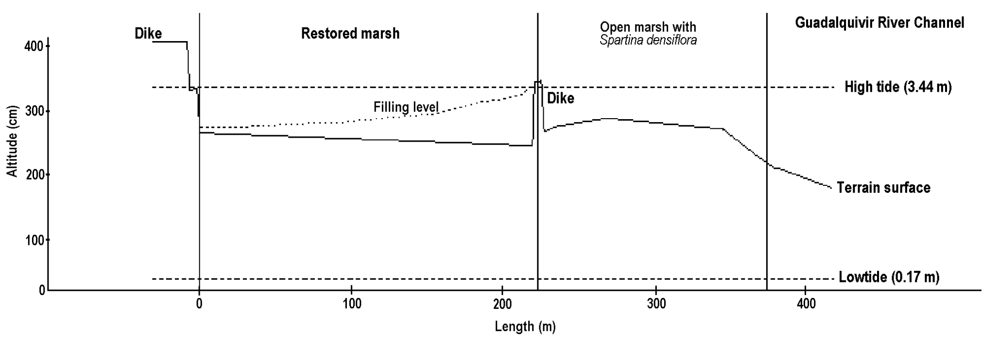


Figure 3. Cross-section of the Algaida Marsh showing the low and the high tide levels.

Wide channels were opened to enable the communication of the marsh with the estuary in order to restore tides and let rain flow out. Channels also offered dispersion paths for active aquatic species and let water masses flow and transport small passive organisms and nutrients to the marsh.

Two approaches have been followed in each of the marsh sectors:

i. Intense intervention with the opening of large channels and ditches, a pond, and controlling weirs (North sector, 22 ha).

ii. Light intervention with the opening of channels to the estuary, inducing geomorphological processes that will eventually rebuild a dendritic-type marsh (South sector, 30 ha).

In order to better control flooding events and the undesirable effects of a river flood, the ancient outer dikes plus a more recent one separating North and South sectors were retained. When medium or low tides occur, water flow to the marsh will take place through the estuary channels only. Outer dikes were leveled off at a height of 2.85 m limiting to 17% the high tides that flow over them. The design included two 30 m long dike sections lowered to a height of 2.70 m, their surfaces protected with gravel, letting almost one half (48%) of the high tides to flow over dike sections for some time. Water entering over the dikes will flow out largely through channels, helping reduce sedimentation there.

North sector (22 ha)

The design established a bracket of floodable surface with a maximum of about 90% under the highest annual tide (3.30 m) and a minimum preserving a permanently flooded surface of about 10%. Two more conditions were imposed: firstly, the water depth over at least half of the minimum flooded area should be 1 m or more and secondly, the whole water volume should be able to drain out by gravity.

This design implies the use of weirs and limits the excavation of channel bottoms not deeper than low tide levels. Also it regulates the turn over of water bodies because in periods of low amplitude tides, for a number of days there will be no net inflow from the estuary to the restored area, depending on weir level.

To provide for the two above-mentioned conditions, wide channels were excavated towards the estuary, with large weirs and a network of inner ditches some wide and deep and many more shallow and narrow, were constructed, helping water circulate easily and permitting organisms to reach the mosaic of habitats.

Estuary channels were 11 m wide on the surface and 5 m wide at the bottom. The bottom level was higher than the lowest tides. Weirs were built 12 m wide and with a fixed level at a height of 2.35 m, a level surpassed by 67% of high tides in the year. Weirs were in the shape of a small dam constructed with a mixture of gravel and local silt, protected with a textile coating and covered again with graded gravel. The diameter of the pieces of gravel in the outer layer were 20-30 cm for added surface stability. Although water flow speed at weirs was estimated at 0.5-1 m/sec (max 1.13 m/sec), the area can suffer from river flooding when higher flow speeds occur.

The two weirs of the north sector form a cross section of 5 m^2 for water inflow before outer dikes are exceeded. This was calculated to suffice for the filling of marsh surface and to compensate for water loses during periods of low amplitude tides not entering through weirs. A hydraulic model was applied (AYESA 2000) to compute water levels inside the north sector under the extreme periods of the year when maximum and minimum tide amplitudes occur. This was needed to adjust the volume-area curve, in order to prevent important water level differences from building up between the estuary and the marsh, which may induce a larger hydraulic charge and a faster water flow at weirs than that designed.

Inside the North sector, 2,060 m of channels and ditches and a pond were excavated, retaining a volume of permanent water of 30,000 m^3 covering, 3 ha (13.6%). Under the highest tide of the year the volume may increase to 200,000 m^3. An average value of 113,000 m^3 is representative of the water exchange per average tide cycle.

Sand and silt excavated from channels was piled up to build three islands of approximately elliptic shape and surfaces of 9,580, 8,784 and 5,924 m^2. Island surfaces represent 11% of the north sector area. Borders have been built to encompass gradients ranging from gentle (1:10) to sharp (1:3) and to ensure wide or narrow ecotone transitions to the floodable area.

South sector (30 ha)

The underlying design philosophy for this sector was to reduce intervention to a minimum, letting geomorphological processes to operate, transforming the initial flat surface into a dendritic type marsh. To foster the transformation, three channels were opened to the estuary with smaller dimensions than those of the north sector (3 m wide with bottom at a height of 1.95 m). No weirs were built,allowing erosion to freely excavate channels. Towards the inner part of the South sector, estuary channels were continued for 25 to 75 m letting inflow water easily expand with tides.

In the south sector water enters with 98% of the annual tides, although volumes are smaller than in the north sector because there are no inner ditches or ponds. Minimum flooded area is about 5%. Maximum flooding covers 95%. Another difference is the absence of islands of piled up material. There is, however, an area of about 1.5 ha (5% of the surface) with a thicket of 17 large tamarisks, each on a large sandy mound. This area was protected with an earthen dike to prevent tides from coming into contact with the surface, allowing rainwater to collect in it, thus favoring a freshwater habitat. In a sense, the protected area behaves like an island in the marsh or a minute archipelago, surrounded by freshwater.

Step Two: Habitat diversification and soil addition

As stated above, habitat diversity was the key feature of the Algaida restoration, and substrates were modified by adding sands or gravel in terrestrial (non-floodable) and aquatic environments (Figure 4).

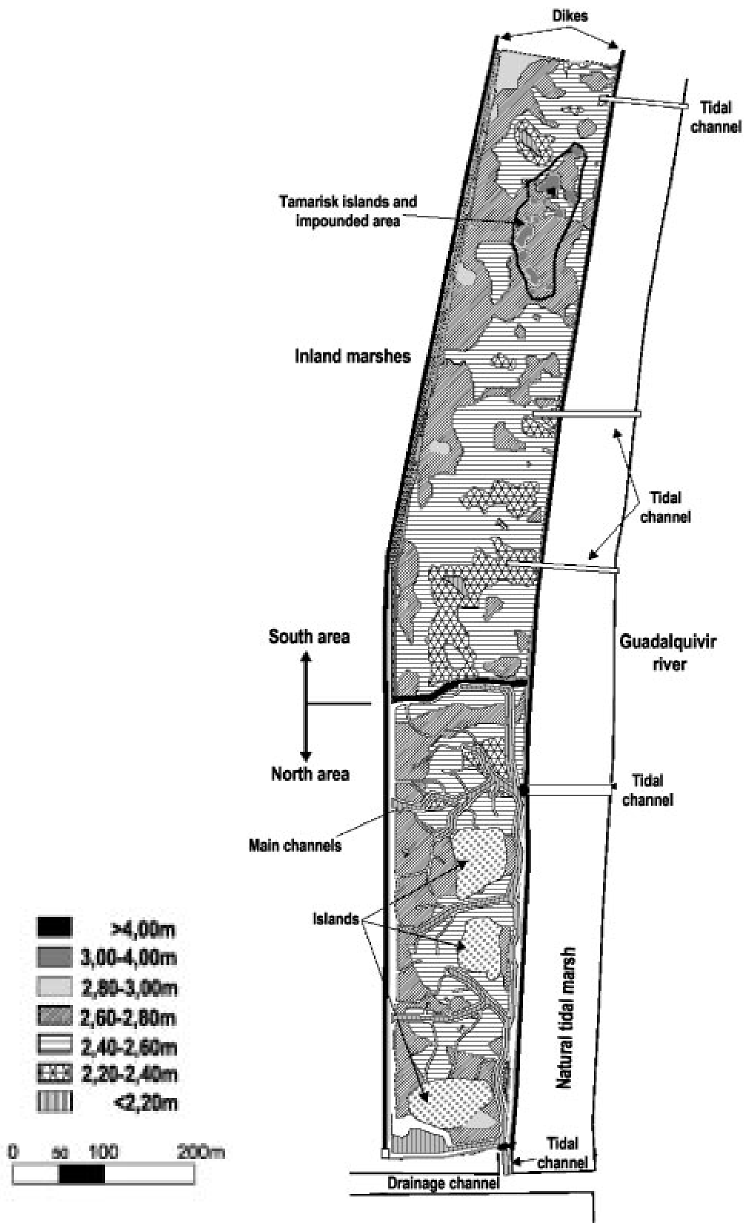
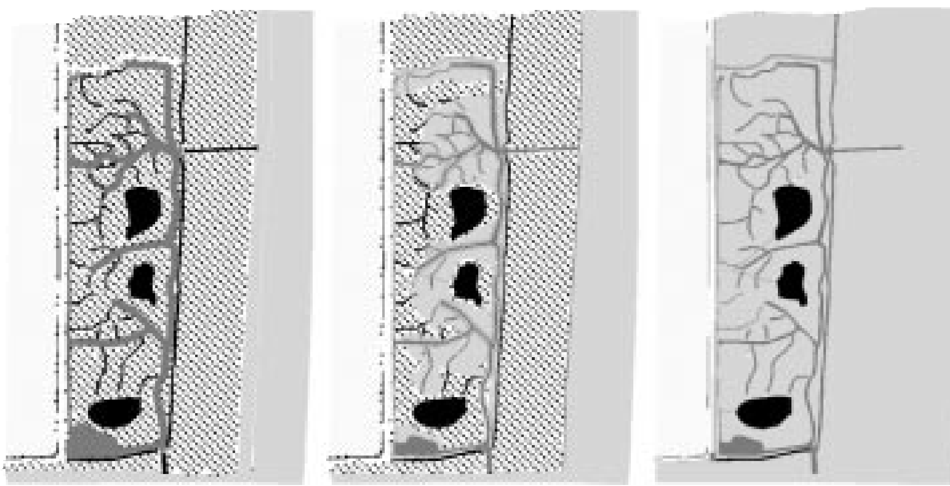


Figure 4. Elevation pattern in the Algaida Marsh with dikes, canals and ditches.

In the north sector two of the islands were covered with a 10-20 cm thick layer of sand. On the three islands, dike sections 50 cm high, 1 m wide and 20 m long were built, offering a drier habitat (intense runoff) and a perch for birds. Gravel bands 2 m wide and 10-20 cm deep were laid, crossing topographic variation from channels to islands. Small mounds, 1 to 5 m across, of gravel (of acid igneous rocks, compact and porous limestone), were added to offer small surfaces of diverse substrate and texture, with varied flooding duration. A part of the pond bottom was also covered with sand at two depths, offering sandy substrates for benthic organisms. Pine trunks (5- 7 m high) were erected on small sandy mounds (6 m across) to serve as perches for birds.

Hydrology is the dominant factor that will determine the zonation of animal and plant species and other biological and physical characteristics of floodable areas (Broome and Craft 2000). In Algaida, marsh zonation was controlled by variations in elevation and slope, that interact with the tidal regime to determine the real extent of the intertidal zone and the depth and duration of flooding (Figure 5).



Minimum tides | Medium tides | Maximum tides

Figure 5. Impounding pattern in restored salt tidal marsh.

In the North sector, a floodable surface covering 17 ha (75.3%) was diversified in two ways: by shaping it into a gentle slope and adding variation to its surface.

The slope has a gradient of 0.5%, rising from channels to island borders and thus causing some surfaces to be inundated more often and longer than others. The “upper marsh” is flooded for a series of 3 days every 15 days; it covers 6 ha (27.3%). The “intermediate marsh” is flooded for a series of 7 days every 7 days; it covers 7 ha

(31.8%). The lower marsh is flooded for a series of 10 days every 5 days; it covers 4 ha (18.2%). Due to their elevations, tide water fails to enter the marsh during two series of 5 days per month.

Floodable surface variation was induced by digging hollows or small shallow pockets (1x2, 2x2 m), where tide water was retained for longer than in their surroundings. Also, small ditches (50x50 cm sections) were opened, crossing the surface to help redistribute water flow and add diversity to the aquatic environments.

The described wetland habitats greatly favor aquatic birds and fish. They also sustain high-productivity in terms of various types of biomass which facilitates the transfer of food from one trophic level to the next. Water persistence in the pond and the channels attracts estuarine fish, which in turn attract fishing birds. Channels offer adequate ground for the reproduction of estuarine species, although water salinity cannot be controlled.

In the South sector, 95% of the surface corresponds to floodable marsh largely dominated (55%) by the upper marsh regime of the north sector. The intermediate marsh represents a further 20% and the lower marsh another 20%, although flooding duration is shorter than in the north sector because of smaller estuary channel cross sections. As erosion proceeds in them, flooding duration shall increase.

The approach to increase biological diversity in the area by creating a diverse habitat mosaic is based on the existence of biological bridges to organisms in dispersion. Birds, a few mammals, many insects, and several plant species may easily to reach to any surface in the Algaida marsh. Others need a pathway for access to favorable habitats and this is particularly important to aquatic species. If dispersion was possible succession would allow communities to build up in any habitat type.

“Self-design” denotes the process in ecosystem development whereby natural mechanisms contribute to species introduction and selection (Mitsch 2000). If an ecosystem is open to allow sowing of enough species propagules, the system will adjust the biological community to existing conditions. The design clue to favoring self-design is to open the ecosystem to newcomers.

In the North sector, the design included the interconnection of channels and ditches so that any floodable surface was available to colonization under passive transport by tidal flow. In the south sector where intervention was kept to a minimum, ditches were opened so as to connect all depressions with the main estuary channels.

Terrestrial organisms not dispersed by air have more restricted access to the islands. Floating fruits, seeds, stolons, or other plant organs adrift on the estuary easily enter the channels and reach the islands, and maybe some invertebrates can be dispersed with them. Other species are known to be dispersed by birds in plant materials as they build nests or in soil particles stuck to them.

In order to favor plant growth, the sand brought to the north sector islands resulted in a soil profile about 50 cm deep, rich in seeds, stolons and soil fauna. No vertebrates (lizards, snakes, rabbits, or rodents) of the fauna associated to the source area were ferried to the islands.

Monitoring

The restoration incorporated a new design based on the diversification of habitats and the induction of self-restoring geomorphological processes. In order to evaluate design and compare the Algaida Marsh with others in the Natural Park of Doñana, a monitoring program has been established, covering vegetation, fauna and geomorphology. To that purpose, some ancillary works were carried out during restoration.

Topography bases were fixed and permanent rods with scales to read water level were installed. Access paths and bridges over channels were built to ease the sampling in the islands. Iron rods were placed at fixed distances and concrete 1x1 m plates were deployed on the channel bottom in order to record geomorphological changes and to estimate erosion or sedimentation rates.

Ancillary works included: A tamarisk plantation to create a green screen along the tracks, a bird watching observatory, signs for visitors and maintenance work on weirs.

After completing the works, channels were opened and estuary waters flooded the restored Algaida Marsh in November 2000. In two tidal cycles the area was filled, birds arrived and terns started preying on the early fish. This marked the onset of monitoring programs, which focused on hydrology, fauna and vegetation.

Water levels inside and outside the restored area and water velocity in weirs and channels during tide cycles were studied to check the accuracy of models. After this validation, exceptional conditions such as Guadalquivir river inundation and very intense precipitation episodes were carefully recorded. As a consequence, the weir longitudinal profile was improved and dikes were reinforced in a few sections early in 2001.

The monitoring of water bodies included salinity, nutrients, temperature profiles and flow rates at channels and pond, and sediment load.

The biota of aquatic environments has been monitored at the various habitats with plankton and benthos sampling. Fish and large crustacean (Decapoda) catches with nets in channels and pond have been carried out in the North sector.

Bird counts were taken at fortnight to monthly intervals at both sectors and adjoining marshes and riverbanks.

Inventories on the vegetation of the area before and during the restoration works were carried out. After the tidal regime was established, inventories of permanent plots in the various types of habitats were taken. The north sector islands and the tamarisk mounds in the south sector were inventoried for vegetation structure and to estimate primary productivity. Some experiments on manipulation of plant cover to control community dominance and to limit the implantation of *Spartina densiflora* have been started. This species has been displacing native tidal marsh angiosperms in the Guadalquivir estuary, and now dominates large stretches of the banks in almost pure stands.

The monitoring program is conducted by the Department of Ecology in the framework of the Algaida Project, a research initiative of the University of Seville, in connection with the Natural Park of Doñana and the Ministry of the Environment. The early reports are being published (Gallego Fernández and García Novo 2002 a, b).

First results

The self-revegetation of the sandy islands started after the 2000 autumn rains with the appearance of annual species such as *Spergularia salina*, *Parapholis incurva*, *Juncus buffonius*, *Sphenopus divaricatus*, *Frankenia laevis*, *Polypogon maritimus* ssp. *maritimus* and *Hordeum marinum*. Strong differences are noticeable between the dense cover of sandy islands (which received seed containing soil) and the sparse vegetation of the silt island (with drifting seeds only). After the 2001 autumn rains, the sandy islands again developed a much denser plant cover than other areas. The incorporation of a seed containing soil induces a marked difference in the early stages of succession.

The floodable surface of the marsh has shown a marked revegetation (Figure 6) which is more intense in the upper reaches where a fair amount of drift materials was deposited during high tides and high river waters.

Figure 6. Revegetation in the Algaida Marsh

Vegetation composition is formed by Chenopodiaceae species belonging to *Salicornia*, *Atriplex*, *Salsola*, *Suaeda*. *Salicornia europaea* and *Sarcocornia perennis* with S. fruticosa dominating plant cover and *Spartina densiflora* presenting point stands. In addition, species like *Mesembryanthemum nodiflorum*, *Spergularia salina*, *Hordeum leporinum*, *Parapholis incurva*, *Plantago coronopus*, *P. crassifolia* and *Cotula coronopifolia* have been recorded.

Plant diversity is highest in the sandy islands that account for about one half of the recorded plant species. Tamarisk mounds (where soil profile was preserved) offered highest diversity with 35 species recorded. Sandy islands hosted 30 species in marked contrast to silt islands (9 species). The high marsh and the marsh fringe around islands show high diversity values with 26 species. A different set of 26 taxa (with many Fabaceae and Asteraceae) are restricted to the sandy islands, emphasizing the importance of habitat diversification and accessibility to mosaic pieces.

The low elevation in the floodable plain or the channel borders may house a few perennials: *Scirpus maritimus*, *Phragmites australis*, and the occasional presence of succulent Chenopodiaceae and *Spartina densiflora*.

In channels, large floating masses of *Ruppia maritima* grow.

Several species of fishes has been registered in the main channels, namely *Atherina boyeri*, *Fundulus heteroclitus*, *Gambusia holbrooki*, *Pomatoschistus minutus*, *Mugil cephalus*, *M. capito*, *M. aurata*, *Anguilla anguilla* and two species of crabs, namely *Carcinus maenas* and *Uca tangeri*.

Birds coming from the nearby Doñana National Park Marsh immediately started exploiting food resources or using the area for perching or to rest. In winter and spring four species nested. In monitoring censuses 65 bird species were recorded during the first year; during the second year 12 species were added to the count.

In the lower marsh, at any one time limicolous species gather. Dunlin (*Calidris alpina*), redshank (*Tringa totanus*), greenshank (*T. nebularia*), little ringed plover (*Charadrius dubius*), ringed plover (*C. hiaticula*), kentish plover (*C. alexandrinus*), grey plover (*Puvialis squatarola*) and black and bar-tailed godwit (*Limosa limosa*, *L. lapponica*) may be encountered. In shallow waters, avocet (*Recurvirostra avosetta*) and black-winged stilt (*Himantopus himantopus*) are abundant. The spoonbill (*Platalea leucorodia*), flamingo (*Phoenicopterus ruber*), great white egret (*Egretta alba*) or white stork (*Ciconia ciconia*) also visit the area. Three individuals of the rare black stork (*Ciconia nigra*) were observed feeding on the Algaida Marsh for weeks. In deeper waters and during the high tides, there are diverse waterfowl: mallard (*Anas platyrhynchos*), red crested pochard (*Netta rufina*), pochard (*Aythya ferina*), wigeon (*Anas penelope*) and shoveler (*Anas clypeata*). In the border or inner channels or close to the weirs one or two individuals may be seen of grey and purple herons (*Ardea cinerea*, *A. purpurea*), little egret (*Egretta garzetta*) or grand cormorant (*Phalacrocorax carbo*). Several terns

(*Gelochelidon nilotica*, *Sterna caspia*, *S. sandvicensis*, *S. albifrons*) may be seen fishing in the deeper waters. Large numbers of little tern (*Sterna albifrons*) gather during low tides in the area for rest.

Marsh harrier (*Circus aeruginosus*) and black and red kites (*Milvus migrans*, *M. milvus*) have been recorded as flying over the area. Occasionally other birds have been recordered, such as the Imperial eagle (*Aquila adalberti*), the pilgrim falcon (*Falco peregrinus*) or the griffon vulture (*Gyps fulvus*), but passing birds will probably comprise more than 200 species. This is because in the Doñana Park the long-term (50 years) monitoring of birds has yielded a figure of *ca*. 380 species (García et al. 2000).

One unexpected result of restoration was the presence of three ospreys (*Pandion hialetus*) during winter. Ospreys used to fish in the channels using the planted trees as perch for feeding and resting. This large bird of prey is endangered in the Iberian peninsula due to habitat loss and does not reproduce in Andalusia.

Belonging to a very different group, the aquatic moss *Riella helicophylla* with few locations in National or Natural Parks, was recorded in the tamarisk precinct of the South sector, in freshwater temporary ponds, where it formed a large colony.

The last two examples show that if habitats were made available and suitable access was provided, many interesting species could enrich restored areas. The monitoring program is documenting it in Doñana, in some detail.

The Algaida Marsh is in the process of recovering its lost biodiversity.

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EPILOGUE

The overexploitation of water resources and the need for agricultural land resulted in the loss of numerous wetlands in the Mediterranean basin. In addition, the pollution and degradation of most of the remaining wetlands indicate the importance of restoration. Nowadays, most of the restoration projects are located at the north part of Mediterranean basin, while a few restoration attempts may also be found in South Mediterranean countries. However the scientific base of these activities is not always strong enough and the results in certain cases are still unclear.

Structural changes of the wetland directly and indirectly affect the degree of performance of wetland functions and thus the biota of the ecosystem. Although for several decades the species oriented restoration was dominant in the region, today it has been realized that the orientation should wider. In all cases hydrology is the key to restoration, although the soil substrate of the wetland influences the final outcome.

The restoration of wetland functions related to hydrology is considered to be of primary importance to the Mediterranean. Because of the increasing water demand in the region and the alteration of global climate, wetland restoration is called to play a leading role in the management of water resources. The hydrologic pathways transport energy and matter to and from wetlands, thus the understanding of hydrology is fundamental to the ecology and hence restoration of wetlands. Furthermore, the manipulation of the biochemical processes of wetland soils is part of the emerging technology of restoration. The contribution of the watershed's soils in controlling the background nutrient levels of the wetlands is well recognized. Sediments proved to be the memory of wetland ecosystems and their study before any restoration project is considered to be more than useful and necessary.

There is insufficient information on the plant communities of Mediterranean wetlands although vegetation is the basis of all food webs. Descriptive studies are needed as well as studies documenting the response of plant communities and species to stresses arising from the water regime and the salt concentration in the water and in the substrate. The potential of wetland plants to improve the quality of water entering natural wetlands has so far little exploited in real situations in the Mediterranean.

The restoration of wetland functions in the Mediterranean is a challenging new task. The Spanish experience in restoration revealed that biological and hydrological functions are most difficult to restore in coastal wetlands. Also, due to the seasonal lack of water, salinization problems are common in the region and require proper solutions. In the case of Mavrouda wetland in Greece, the restoration scientists had to cope with the salinization problems in order to come up with a sustainable restoration solution of wetland functions.

To introduce one more difficult parameter concerning the restoration of Mediterranean wetlands, it should be emphasized that wetlands are not isolated ecosystems in the landscape. Usually found at the lowest sites of a watershed, wetlands strongly affect and are affected by the surrounding ecosystems, whether natural or man-made, and their management. The Axios river in Greece is a characteristic case where management interventions at watershed level and especially of the agroecosystems, are necessary for the restoration of the river, and the river-dependent wetlands.

Ecosystem restoration is the interface between research and application. One must understand the basic functioning of systems before he can manage or restore them. Applied science is an extension of basic science and totally grounded on it. Considering that hydrological fluctuations promote ecosystem resilience and longevity, it is obvious that restoration should promote the establishment of ecosystems and not just water reservoirs. Although the general principles of restoration theory may be used as a basis for the restoration of Mediterranean wetlands, the detailed restoration actions and the final outcome are site specific and depend on the local environmental and socioeconomic constraints. Because of the unique climatic and geomorphological characteristics of the Mediterranean, wetland restoration models and experiences from other climate zones may not apply untested in the region.

There is a considerable number of skilled scientists in the Mediterranean countries who can address the task of wetland restoration. Restoration scientists should focus on the functioning of wetlands, as it results from the structure of the ecosystem and the interactions with the wider environment. The dynamics of Mediterranean wetlands should be better understood and for this purpose the development of eco-hydrology will be beneficial. Certainly, Mediterranean scientists have a lot to learn from successes and failures in other regions of the world and perhaps there is more to learn from each other's experience since the Mediterranean shows both a diversity and unity of wetland problems and opportunities.

The functional approach at watershed level seems to be the most appropriate for the restoration of Mediterranean wetlands. Restoration scientists should take into account the environmental and socioeconomic constraints of the watershed in order to restore sustainably the desirable wetland functions. The complete restoration of wetland structure is not an option in the Mediterranean. In most cases, the landscape has evolved so much that it is impossible to return to the preimpact hydrological and geomorphological condition. In addition, there is no pristine condition left in the region to use as a reference level in restoration. Rarely do we know what the wetland was like prior to disturbance because the latter is additive and ongoing for centuries.

Even when there is adequate data the existing environmental and socioeconomic conditions do not allow the restoration planners to use a preexisting condition as a reference level. Paleolimnological and historical approaches may be helpful in identifying past conditions. Although restoration must consider structural changes over time as they are reflected in the system's memory that is the sediments, it should not be used to imply a return to a previous state. In terms of restoration planning, the needs of the present, the availability of natural resources, and the environmental constraints suggest the functions that the restored wetland should perform and the ecological characteristics that the wetland should have in order to provide multiple benefits and services within a specific watershed area.

From a socioeconomic point of view, public needs and demands in the region have undergone considerable changes and restored wetlands are called upon to play a rather reformed role in the landscape. Thus, the first question of any restoration project should be for what reason the wetland is wanted. The motives of restoration should be identified from the beginning in order for well-defined restoration objectives to be set.

The planning of a sustainable restoration should not only take into account the environmental constraints of the watershed in order to establish a wetland ecosystem but also should be able to fulfil the expectations of local communities concerning the benefits and services that the wetland will provide. In other words, the ecological characteristics of the wetland should reflect the needs of local society in terms of environmental stewardship and socioeconomic development as they are today and moreover as they are expected to evolve in the future.

The evaluation of restoration success should be prescribed in all restoration studies and the criteria must be well defined.

The success of a restoration project does not only depend on the wide social acceptance of the project and on ecologically and technically correct measures. Success is also requires the formation of an effective scheme to manage the wetland after the restoration interventions. This factor is particularly important for projects re-creating a wetland which had completely been drained for many decades. The management of such wetlands large funds for monitoring and corrective actions and an efficient administrative structure.

SUBJECT INDEX

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