Effect of the river disturbance regime on floodplain structure and organic matter accretion at the Middle Ebro River

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This thesis is based on the following articles, which are referred to in the text by their Roman numerals:

I. Hydrologic and land-use change influence landscape diversity in the Ebro River (NE Spain) *Submitted to Landscape Ecology.*

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CONTENTS

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ALVARITO

ABSTRACT

One reach in the Middle Ebro River (NE Spain) was selected to evaluate the influence of the river disturbance regime over floodplain structure and organic matter accretion in the context of strong and repeated anthropogenic disturbances. Several spatial and temporal scales were considered for the analyses. Based on our results, we aimed to assess the ecological status of the study reach, suggesting guidelines to rehabilitate the river-floodplain system towards a more natural functioning. To that end, we considered the study reach as the larger spatial unit. For this scale, the distribution and dynamics of different landscape ecotopes was selected, in order to asses the trends during the last century. To evaluate the individual effect of different magnitude floods, we selected the hydrogeochemical features and dynamics of surface waters. Secondly, we selected an array of individual riparian wetlands to asses the effect of the river disturbance regime in a smaller spatial scale. To evaluate the effect at decadal time scale, understory diversity estimates were employed, whereas the structure of flooded top-sediment (0-3 cm) was used to study the influence at annual time scale. Finally, organic matter accumulation at floodplain substrates was selected as example of ecosystem function. To evaluate current patterns, we examined the top-soil (0-10 cm) on intermittently flooded habitats. Moreover, modifications of these patterns during the last century were also studied using sediment cores.

According to our results, we can conclude that the river disturbance regime is the main factor explaining the floodplain structure and organic matter accretion patterns in the Middle Ebro River. As a result, human alterations on this regime were clearly reflected in our analyses. Such modifications exerted a direct influence on the elements of the systems which respond to river-floodplain interactions at decadal time scale (landscape, riparian understory, organic matter accretion patterns). Landscape and riparian wetland heterogeneity have markedly decreased, and mature successional stages dominate both riparian forest and aquatic habitats. Wit regards to organic matter, autogenous inputs have become to dominate over river sedimentation, what diminishes the potential of floodplain areas as carbon and nitrogen sinks. Furthermore, modifications of the river disturbance regime have to be taken into account when interpreting the results of elements responding to hydrological connectivity at shorter time scales (hydrogeochemistry, sediment composition, organic matter accretion at the top-soil of riparian forest). A diverse array of wetlands types, in terms of hydrogeochemistry and sediment composition, was found at the study reach. However, the portion of floodplain area presenting such heterogeneity is scarce, and mid-range floods (1.5 y) promote a rapid homogenization of the riverscape. With regards to organic matter at the top-soil, autochthonous inputs dominate if older habitats (>60 y) old, which present the higher budgets, whereas allochthonous does at younger patches, with lower organic matter budgets. When interpreting those results, however, it should be considered the vast extent of agricultural fields and poplar groves, about 75 % of the considered floodplain, which presented low budgets despite they are isolated from river sediments inputs.

In the light of the previous analyses, it seems that ecological restoration is urgently required at the Middle Ebro Floodplains. Basin, reach and site scale projects should be implemented to rehabilitate system components examined in this thesis. To accomplish that, periodic economic investments will be required because self-sustained restoration seems neither possible nor realistic under the current Ebro basin management.

RESUMEN

Con el objetivo de evaluar la influencia del régimen de alteración fluvial tanto sobre la estructura de las llanuras de inundación como sobre la acumulación de materia orgánica en el sustrato, todo ello en un contexto de fuerte alteración antrópica, se seleccionó una sección del tramo medio del Río Ebro, considerándo múltiples escalas espacio-temporales para el estudio. Además, se evaluó el estado ecológico del tramo de estudio, sugiriendo una serie de propuestas para rehabilitar el sistema formado por el río y su llanura de inundación con el objetivo de recuperar un funcionamiento más naturalizado. Para ello, consideramos el tramo de estudio como la escala espacial más grande. A esta escala, la distribución y dinámica de los distintos ecotopos fue seleccionada con el fin de analizar las tendencias durante el último siglo. Además, se estudió las características y dinámicas hidrogeoquímicas en las aguas superficiales con el propósito de evaluar por separado el efecto de riadas de diferente magnitud. En segundo lugar, seleccionamos un conjunto de humedales riparios para evaluar el efecto del régimen de alteración fluvial a una escala espacial más reducida. La diversidad del sotobosque ripario fue seleccionada para analizar dicho efecto a escala temporal de décadas, mientras que la estructura del sedimento superficial de zonas inundadas fue empleada para ver la influencia a escala temporal de años. Finalmente, la acumulación de materia orgánica en los sustratos de la llanura de inundación fue seleccionada como ejemplo de función ejercida por este tipo de ecosistemas. Para evaluar los patrones actuales, se estudió el suelo superficial (0-10 cm) de hábitats inundables. Además, los cambios que durante el último siglo se han producido en susodichos patrones fueron estudiados a través del análisis de testigos de sedimento.

De acuerdo con nuestros resultados, se puede concluir que el régimen de alteración fluvial es el factor principal a la hora de explicar tanto la estructura como la acumulación de materia orgánica en las llanuras de inundación del tamo medio del río Ebro. Por ello, cualquiera de las alteraciones antrópicas que este régimen ha sufrido se ve reflejada en estas comunidades. Tales modificaciones ejercieron un efecto directo sobre los elementos del sistema que responden a las interacciones que se producen, a escala de décadas, entre el río y su llanura de inundación (paisaje, sotobosque ripario, patrones de acumulación de materia orgánica). La heterogeneidad tanto de paisaje como de humedales riparios ha descendido de una manera notable. Las etapas de sucesión maduras dominan tanto los hábitats acuáticos como los de bosque ripario. En lo que respecta a la material orgánica, los aportes endógenos han llegado progresivamente a dominar sobre los sedimentos fluviales, lo cual disminuye el potencial de las llanuras de inundación como sumideros de carbono y nitrógeno. Además, las modificaciones del régimen de alteración fluvial se tienen que tener en cuenta a la hora de interpretar los resultados de los elementos del sistema que responden a la conectividad hidrológica en escalas temporales mas reducidas (hidrogeoquímica, sedimento inundado superficial, acumulación de material orgánica en el sustrato superficial del bosque ripario) En el área de estudio podemos encontrar un diverso conjunto de tipos de humedales riparios, en cuanto a hidrogeoquímica y composición del sedimento se refiere. Sin embargo, la proporción de la llanura de inundación que presenta esta heterogeneidad es escasa, y las riadas de magnitud intermedia (1.5 anos) homogenizan el conjunto de humedales riparios de manera muy rápida. En lo que se refiere a materia orgánica en el sustrato superficial, los aportes endógenos dominan en los hábitats mas maduros (>60 años), los cuales presentan las cantidades mas elevadas, mientras que los exógenos lo hacen en los hábitats mas jóvenes, que presentan las cantidades de materia orgánica mas bajas. A la hora de interpretar estos resultados, sin embargo, se debe de considerar la gran extensión de

campos y plantaciones comerciales de chopo (alrededor del 75 %) de la llanura de inundación, los cuales tienen cantidades bajas de materia orgánica aun cuando permanecen aislados de los aportes de sedimentos fluviales.

Teniendo en cuenta estos resultados, parece clara la necesidad de desarrollar una restauración ecológica de las llanuras de inundación del Ebro Medio. Se deberían de implementar proyectos a escala de Cuenca, tramo y hábitat si se quieren rehabilitar aquellos componentes que han sido objeto de análisis en esta tesis. Para ello, se requerirán inversiones económicas periódicas ya que una restauración auto-sostenida por la dinámica fluvial no parece ni posible ni realista bajo la actual gestión de la Cuenca del Ebro.

SUMMARY

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1. Introduction

1.1 Floodplain benefits: the need of a knowledge-based restoration

Floodplains are ecotones between terrestrial and aquatic systems that are subject to recurring flooding. Under a basin perspective, those are the areas covered by stream transported sediment that was deposited in or near a stream channel. The importance of those ecosystems greatly exceeds their extent because of the intricate links between terrestrial and aquatic ecosystems (Pinay et al., 1992). Floodplains exchanges water, sediment, organic debris and chemicals with surrounding ecosystems as rivers, aquifers and uplands. In other words, those areas receive and manufacture energy and mass, providing habitats and holding physical, chemical and biological processes (Amoros and Petts, 1993; Mistch and Gosselink, 2000). Due to its natural functions, floodplains provide valuable goods and services to humankind as gas regulation, disturbance regulation, water regulation, water supply, waster treatment, habitat diversity, food production, raw materials, recreation or cultural services. The estimate worldwide value of the services provided by floodplains is US$ 3920 * 10^9 yr $^{-1}$. In total, floodplains contribute about 10% of the total flow value attributed to world ecosystems, although they cover only 0.3% of the total world surface (Constanza et al., 1997).

Such wide variety of services provided to the humankind belongs from the fact that floodplains are among the most biologically productive and diverse ecosystems on earth. However, floodplains are also among the most threatened ecosystems (Olson and Dinerstein, 1998; Tockner and Standford, 2002). Floodplains are particularly sensitive to environmental change because they integrate and reflect alterations at different spatial scales (Naiman and Decamps, 1997; Naiman et al., 2002). Reclamation, i.e. elimination of natural habitats from floodplain areas, and modification of the hydrologic regime are the major alterations over floodplains in lowland rivers (Buijse et al., 2002) as the Middle Ebro (NE Spain). Most of the world´s 79 large river-floodplain systems have been altered by human activities (Sparks, 1995). In Europe, between about 60 and 90 % of the entire riparian corridor has been transformed to cropland and/or is urbanized (Ravenga et al., 1998). Moreover, in European and North American rivers, the relationship between human population density and occupation of the riparian corridor is logarithmic, due to floodplains are highly "developed" even at a low population density (Tockner and Stanford, 2002). With regards to hydrology, the majority of large temperate rivers have been strongly regulated during the past few centuries (Petts et al., 1989). In North America, Europe and the former Soviet Union, 71 % of large rivers (premanipulation mean annual discharge > 350 m^3 s^{-1}) are affected by dams and reservoirs, interbasin diversion and water abstraction (Dynesius and Nilsson, 1994).

As a result, the degradation of rivers and floodplains is increasingly being recognised as a crucial political issue with socio-economic repercussions, particularly in North America and Europe (Naiman et al., 1995). However, examples of river-floodplain restoration are few, recent and most of them have focussed narrowly on permanent aquatic habitats, with only few including the riparian zone (Schiemer, 1995; Brookes, Baker and Redmond, 1996; Galat et al., 1998; Toth et al., 1998; Adams and Perrow, 1995; Simons et al., 2001). Furthermore, it has been underscored that a high proportion of restoration project fails probably due to the lack of knowledge of natural conditions (Lockwood and Pimm, 1999). To be successful, conservation and restoration efforts require a strong conceptual foundation and a thorough understanding of natural processes (e.g. Henry and Amoros, 1995; Stanford et al., 1996). As stated by Ward et al. (2001), a proper understanding of processes operating at different spatio-temporal scales is required for an effective restoration of river corridors. Since the river disturbance regime, i.e. floods, is the major function controlling the structure and functionality of floodplain ecosystems (Junk et al. 1989, Tockner at al., 2000), it seems highly appropriated to evaluate, prior to perform any restoration activity, the influence of the disturbance regime over the Middle Ebro floodplains. Natural disturbances caused by low-frequency floods create and maintain a complex mosaic of riparian landforms and associated aquatic and semi-aquatic communities, while hydrological connectivity during floods promotes the exchange of matter and energy between different parts of the river system (Junk et al., 1989; Ward, 1989). Hydrogeomorphic variables therefore establish the physical template and provide constraints under which chemical and biological processes operate (Tabacchi et al., 1998). The complex interactions of processes at different spatio-temporal scales promote a combination of complex gradients of habitat conditions, which can result in high levels of diversity (Ward, 1998; Amoros , 2001). The conservation of these ecosystems depends on the rehabilitation of river-floodplain interactions at several spatio-temporal scales (Henry and Amoros, 1995; Tockner et al., 1998; Hughes et al, 2001; Brunke, 2002)

1.2 The role of river disturbance regime as the primary driver

On the decadal-centurial scale, erosive floods create and maintain a diversity of successional stages that determine the overall complexity of the landscape matrix (Metzger and Décamps, 1997; Galat et al., 1998). Riparian succession tends to drive aquatic environments toward terrestrial landscapes, but erosion and deposition during low-frequency floods truncate those successional pathways. As a result, in a diverse landscape which contains landscape units at every stage of succession, irregular and anticipated, events drive hydrogeomorphological functions and, in general, allow the system to remain stable (Amoros and Wade 1996). As described by Stanford et al. (2005), floodplain elements tend to persist in natural river systems although their spatial distribution shifts over time due to flow-related changes. At young ecotopes, allochthonous inputs of energy and matter control riparian succession, whereas autochthonous processes determine riparian succession at late ecotopes (Corenblit et al., 2007). In the absence of natural disturbances, the floodplain system probably tends toward geographical and temporal uniformity (Tockner et al., 1998). Under these conditions, the riverscape becomes dominated by mature stages since successional pathways can proceed unimpeded and new wetlands are not being created (Ward and Stanford, 1995).

On shorter spatial scales (days to years), the exchange of mass, energy and organisms between the main channel and floodplain during different magnitude events (flood and flow pulse, see Junk et al., 1989 and Tockner et al., 2000) is the primary driver controlling floodplain ecosystems. It has been defined as hydrological connectivity (Amoros and

Bornette, 2002). The type and magnitude of hydrological connectivity is determined in lowland rivers by inputs from the different sub-compartments of the river-floodplain system: Bank seepage from the main channel, vertical recharge during overbank flooding, the extent of the hyporheic corridor, the influence of local alluvial and hillslope aquifers, and inputs from main channel surface. The exchange of nutrients, organic matter, inorganic sediments, organisms, seeds and vegetative material between the main channel and its adjacent floodplains, as well as flow velocities and scouring during floods, depends on the importance of different inputs. It finally shapes the structure and functionality of a given riparian ecotope under specific geomorphological traits, as widely demonstrated by others for aquatic communities, vegetation structure, water chemistry, substrate characteristics, sediment deposition, nutrient cycling, organic matter ex-change or primary productivity (Rostan et al., 1988, Van der Brink et al., 1993; Asselman and Middlekoop, 1995; Heiler et al., 1995; Megonigal et al., 1997; Tockner and Schiemer, 1997, Bornette et al., 1998; Amoros and Bornette, 1999; Hein et al, 1999; Tockner et al., 1999; Malard et al., 2000; Robertson et al., 2001, Amoros and Bornette, 2002; Pinay et al., 2002; Steiger and Gurnell, 2003; Weng et al. 2003, Baker and Vervier, 2004; Hein et al., 2004; Hefting et al., 2004; Lockaby et al., 2005; Noe and Hupp, 2005; Lanhgans and Tockner, 2006).

1.3 Organic matter accretion at floodplain substrates

Studies dealing with the role of floodplains as permanent organic matter sinks are scarce despite the importance of organic carbon and nitrogen accumulation. In riparian ecosystems, nitrogen retention can enhance the quality of surface water (Johnston, 1991; Day et al., 2004; Verhoeven et al., 2006) and, thereby, prevent the eutrophication of aquatic ecosystems downstream. With regards to carbon, enhancing ecosystems as carbon sinks to ameliorate the greenhouse effect has been suggested as a potential extension of international climate treaties (IPCC, 2000). To that end, soil organic matter (SOM) acquires special relevance since conforms a larger and more perdurable pool that OM accumulated at living biomass. SOM includes plat, animal and microbes residues in all stages of decomposition. Such complexity needs to be understood due to the influence of OM quality, i.e. how difficult it is for SOM to be biodegraded (Rovira et al., 2008), on the potential of floodplains as OM sinks. To achieve that, an older and more recalcitrant pool can be differentiated from a younger and more easily decomposable fraction through the acid hydrolysis approach (Paul et al., 2006).

To evaluate C and N accretion at floodplain substrates, a correct understanding of the effect of processes operating at different spatio-temporal scales is required. OM produced by biotic assemblages is incorporated into surface soils in the intervals during floods. In turn, low-organic sediments are deposited over riparian landforms during floods, whereas OM stored in different compartments of the system (aboveground biomass, sediment, litter) is either exported or buried. As pointed by Daniels (2003) the aggradation rate is inversely correlated with soil development. At early successional stages, hydrological connectivity is high and substrates are therefore more inorganic due to high sedimentation rates. Moreover, OM exports occur more often. At late stages, ecotopes are more disconnected from the main channel and their substrates are more organic due to the incorporation to the soil pool of in-situ produced OM.

1.4 The Middle Ebro, a regulated lowland river

In the Ebro River, alterations of river flows and floodplains have disrupted the intensity, frequency, and timing of the natural disturbance regime responsible for maintaining the ecological integrity of these ecosystems. Attracted by its fertility, humans have occupied the floodplain of the Ebro River since before Roman times. However, during the last century the flow regime has been greatly disrupted. Irrigation of lowland areas and abandonment of farm-lands in upland areas took place throughout the 20´s century, dramatically changing both hydrology and sediment transport (García-Ruiz et al., 1995; Ibañez et al., 1996; Beguería et al., 2003; Batalla et al., 2004; Beguería et al., 2006; Lopez-Moreno et al., 2006; Vericat and Batalla., 2006). The construction of dams began in 1913, but most were built between 1940 and 1975. The Ebro River currently has 234 reservoirs impounding 57% of the mean annual runoff (www.chebro.es). The promotion of dam construction, mainly for irrigation purposes, resulted in the accelerated occupation of river margins and massive construction of flood protection structures (Pinilla, 2006). At the Middle Ebro, mean annual runoff declined approximately 30% since 1981. Several contributing factors have been suggested, including progressive increases in evapotranspiration due to higher temperatures, reforestation , water storage and the expansion of irrigated farmlands (Ibanez et al., 1996; Ollero, 2007). In addition, agricultural production since the 1980's has been largely driven by an emphasis on the cultivation of water-hungry crops, such as rice, fruits and vegetables (Frutos et al., 2004). At the study reach, discharge magnitude and variability have markedly decreased since the 80´s (unpublished data), while modern protection structures were built between 1960 and 1980 (Ollero, 1992), and they are reinforced after every large flood. Earlier studies (Regato, 1988) reported that natural vegetation had been strongly modified within at the study reach by such alterations or river-floodplain interactions; this was later confirmed by Castro et al. (2001).

In this context of strong and repeated anthropogenic disturbances, the aim of the thesis was to analyze how the river disturbance regime is related with the ecosystem structure and organic matter accretion at the Middle Ebro Floodplains. To acquire a correct understanding, several spatial (habitat and reach) and temporal (individual floods, annual and decadal) scales were evaluated. Based on our results, we assessed the ecological status of the study reach, suggesting guidelines for restoring river-floodplain interactions at the Middle Ebro River. To accomplish that goal, we selected one relatively free-flowing reach at the Middle Ebro River, where natural riparian habitats at different successional stages and distinct connectivity type can still be found. The study reach includes the Galachos Natural Reserve (Zaragoza).

2. Objectives

The three main objectives of this thesis were:

a) Asses the effect of the river disturbance regime on the structure of habitats at the middle Ebro floodplains at different spatial and temporal scales.
b) Asses the effect of the river disturbance regime on organic matter accretion at the substrates of the Middle Ebro floodplains.
c) Based on the previous analyses, evaluate the ecological status of the study reach and propose restoration guidelines.

To achieve those goals, different components of the river-floodplain system were selected in order to understand processes at different spatio-temporal scales (Fig.1). Specific aims of this thesis are explained in the following sections.

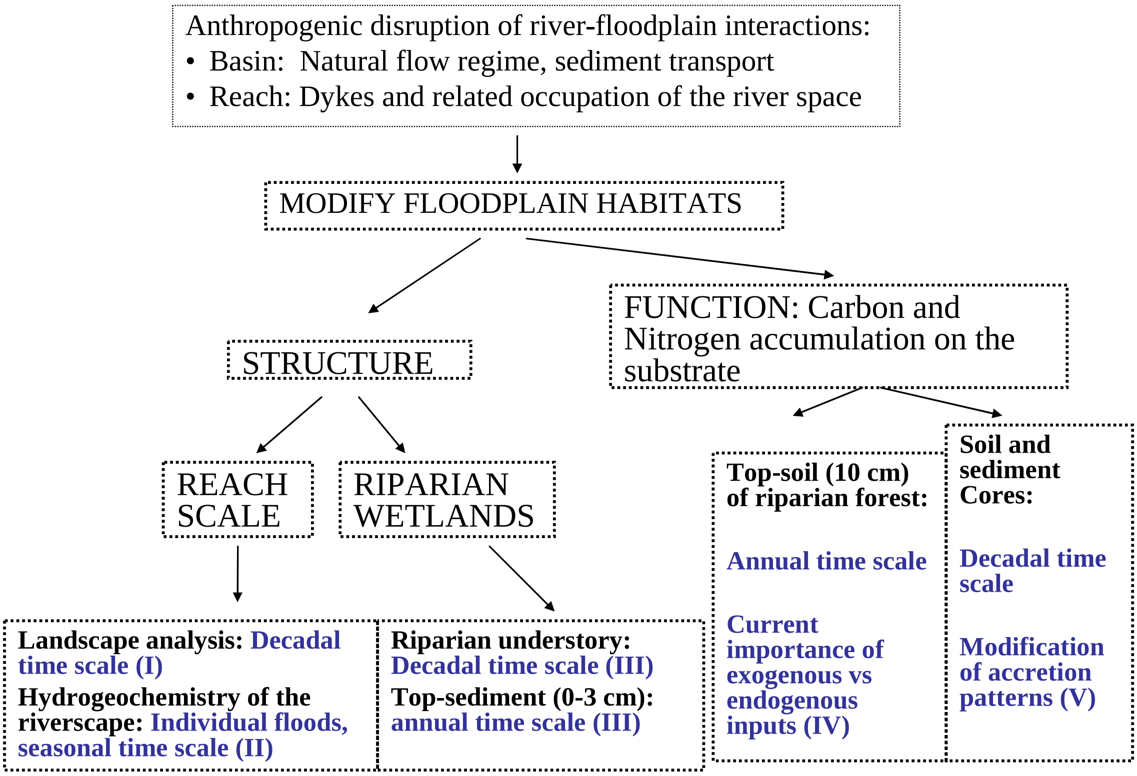

Figure 1. Thesis scheme. Different spatio-temporal scales considered are indicated. Also the elements of the river-floodplain system selected to asses the effect of hydrological connectivity at each scale are displayed

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2.1. River disturbance regime and floodplain structure

Two spatial scales were differentiated. First, we considered the study reach as the larger spatial unit to perform our analyses. For this scale, the distribution and dynamics of different landscape patches was selected to asses the effect of the disturbance regime over floodplain structure during the last century (**study I**). The diversity of landscape units and their spatial distributions in pristine riverine landscapes are the result of complex processes and interactions operating across a wide range of spatio-temporal scales. As a consequence, it has been through the interpretation of sequential landscape patterns that the primary drivers of the riverine landscape dynamic have been inferred in different studies (Hohensinner et al., 2004; Geerling et al., 2006; Whited et al., 2007). The specific objectives of this section were:

a) Examine changes in hydrological and landscape patterns at one selected reach located in the Middle Ebro floodplains.
b) Identify the factors that best explain the natural ecotope succession

To asses the effects of seasonal floods at reach scale, we selected the hydrogeochemical features and dynamics of all permanent and intermittent water bodies at the study reach (**study II**). Several studies have examined the effects of hydrological connectivity on the water chemistry of riparian wetlands on multiple spatial scales (Trémolierés et al., 1993; Heiler et al., 1995; Knowlton & Jones 1997; Malard et al., 2000; Domitrovic, 2003; Hein et al., 2004). The specific objectives of this section were:

a) Identify major hydrogeochemical processes affecting surface water physico-chemistry
b) Define wetland types with regards to their hydrogeochemical features and dynamics

Secondly, we selected a diverse array of riparian wetlands with different geomorphological evolution and hydrological connectivity to asses the natural disturbance regime in a smaller spatial scale (**study III**). We defined riparian wetlands as permanent water bodies and their surrounding areas. Wetland limits were defined by human landscape features (e.g., paths and crops) for the oxbow lakes and artificial ponds. For the outer banks of side channels and backflow channels, limits were demarcated by abrupt topographic changes, whereas the limits of sides adjacent to the riverbank were marked by the point bars associated with the main channel.
To asses the effect of natural disturbance at decadal time scales, understory diversity estimates were employed. This analysis relied on the close relationship of riparian vegetation with wetland topography, thus with wetland successional stage (Ward and Stanford, 1995). To study the effect of natural disturbances at annual time scale, sediment structure was used to infer the hydrological connectivity type. This approach has been previously used in other lowland rivers (Rostan et al., 1987; Tockner and Schiemer, 1997). The specific objectives of this section were:

a) Relate sediment and riparian understory structure with natural disturbances in a variety of riparian wetlands.
b) Assess the ecological status of the study reach.

2.2 River disturbance regime and organic matter accretion

We selected the organic matter accumulation at floodplain substrates as the function to evaluate through this thesis. The major role of riverine floodplains as sinks, sources or transformers of organic matter has been previously highlighted by many (Junk et al., 1989; Walling et al., 1996; Tockner et al., 1999; Robertson et al., 1999; Hein et al., 2003). However, studies dealing with its role as permanent organic matter sinks are scarce. To evaluate current patterns on OM accretion, particularly the importance of autogenous vs. autochthonous inputs, we studied the top-soil on intermittently flooded habitats (**study IV**). The specific objectives of this section were:

a) Evaluate the influence of landform evolution on the accumulation of soil organic matter (SOM) in the topsoil of natural riparian ecosystems
b) Examine, within a floodplain perspective, the role of riparian soils as organic carbon and organic nitrogen sinks.

To evaluate how accretion patterns have been modified during the last century, sediment profiles of four riparian habitats were examined. Historical accretion rates were estimated using Cs-137 dating and aerial photographs (**study V**). The specific objectives of this section were:

a) Analyze the spatio-temporal heterogeneity on organic matter accretion patterns
b) Asses the potential of the middle Ebro floodplains as carbon and nitrogen sinks

2.3 Ecological status of the study reach: Implications for restoration and management

Considering the results obtained in the different sections, we aimed to evaluate the ecological status of the study reach. After that, we propose guidelines for the future management and restoration of the Middle Ebro Floodplains. The specific objectives of this section were:

a) Interpret results of landscape analysis, hydrogeochemical dynamic, vegetation sediment characteristics and organic matter accretion within the study reach context.
b) Propose guidelines at basin, reach and site scale which could match different restoration targets.

3. Study area and methods

The study reach (Fig.2) was located in the Middle Ebro River, NE Spain. This is the largest river in Spain (watershed area = 85,362 km^2, river length = 910 km, average annual discharge to the Mediterranean Sea = 14.442 hm^3/y) and is still geomorphologically active. The river meanders within the middle Ebro River (sinuosity = 139, bank slope = 0,050%), resulting in an average floodplain width of 5 km (Ollero, 1995). Within the study reach, the average monthly discharge is 230 m^3/s and the elevation ranges between 175 m a.s.l. in the river channel to 185 m a.s.l. at the base of the scarp. At the Zaragoza city gauging station (A011, 12 km upstream the study reach), the potential storage capacity is 1.637.19 hm^3, mostly stored by two big dams constructed in 1945 and 1954 respectively, impounding about 50% of the mean annual runnof. At the study reach, the mean annual run-off and the bankfull discharge has declined approximately 30% since 1981. Since 1981, the natural flow regime has been greatly disrupted (Fig.3). Events above the bankfull discharge decreased in magnitude and duration while those below bankfull discharge decreased only in number (Tab.1). Moreover, flood control structures have been built and reinforced every large flood at the study reach since the 60´s (Ollero, 1992).

Agricultural fields and mature ecotopes have increasingly dominated (75 % of floodplain area, unpublished data) over other patch types since 1957 (Fig.2), whereas lateral migration of main channel no longer takes palace since 1981 , promoting mature ecotope stages to dominate. The area flooded by the 10-yr return period flood (3000 m^3/s, 1927-2003) is 2230 ha, although only about 14% of the area is flooded by a river discharge of 1000 m^3/s (0.37 y return period, 1927-2003), and only 4% is flooded by a river discharge of 600 m^3/s (0.14 y return period, 1927-2003). During the last century, the number and extent of permanent water bodies has declined considerably.

In the study **I**, The landscape dynamic was examined using aerial pictures and GIS techniques. Moreover, changes in the natural flow regime and anthropic activities within the river-floodplain system were investigated. In order to explore the relationships between river dynamics and landscape changes, hydrological data were separated and analyzed for three different periods (1927-1957; 1957-1981; 1981-2003). Then, ecotope maps were generated from a set of aerial photographs (1927, 1946, 1957, 1981, 1998 and 2003) to perform a landscape transition analysis using GIS techniques. To explore the relationship between landscape structure and river dynamics, ecotope maps were progressively truncated by increasing the distance to the main channel. For all buffers considered in this study, Fragstats 3.3 (McGarigal and Marks, 1995) was used to calculate the area, and percentage of land occupied by each ecotope category, as well as ecotope diversity using the Shannon Index. To examine ecotope change, transition matrices and maps were produced for each time span using IDRISI Kilimanjaro® (CrossTab).

In the **study II**, surface water samples were collected during two flood events and one low-discharge period to define the hydrogeochemical features and dynamics at the study reach. Water samples were

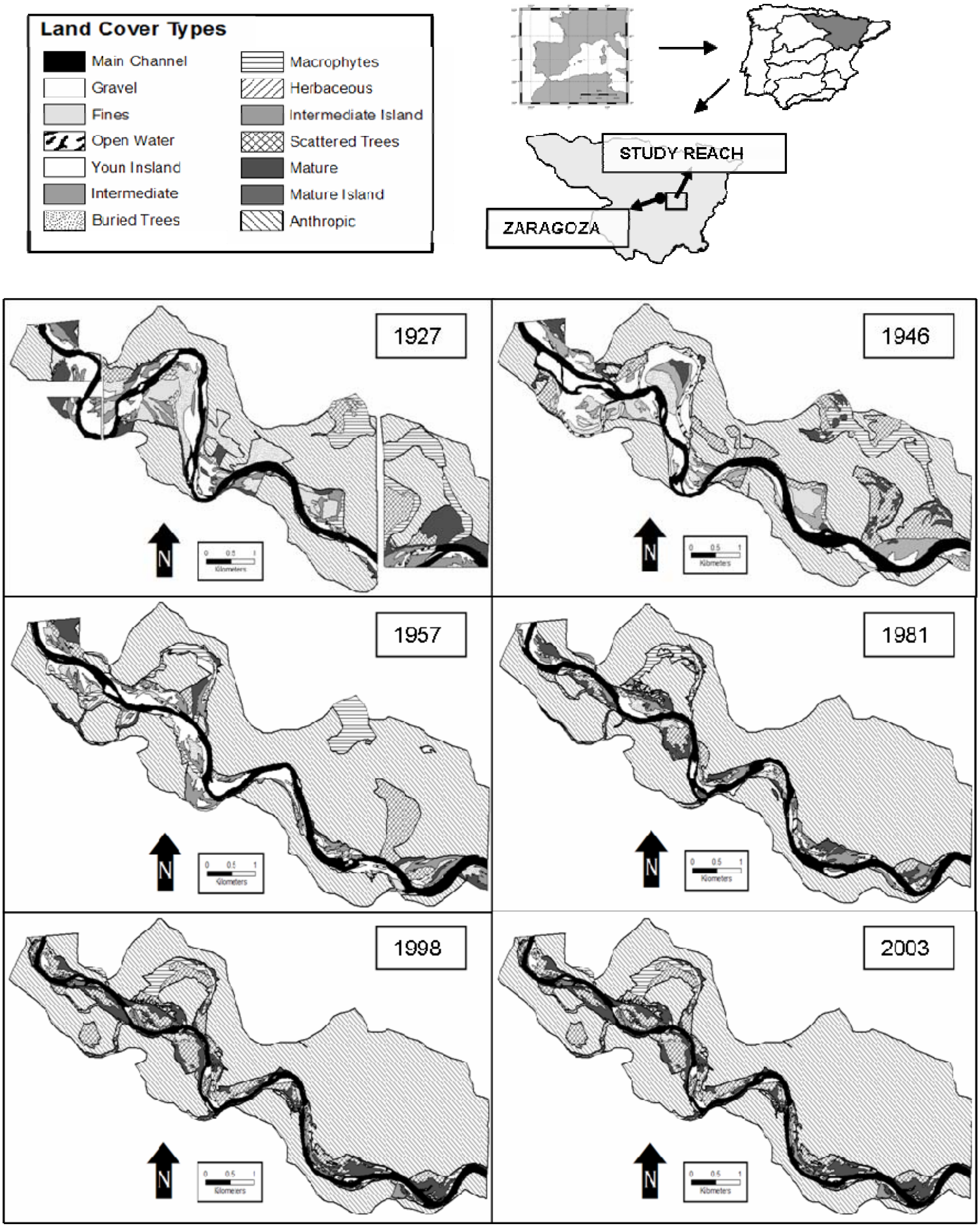

Figure 2. Study area and ecotope maps for six different years. River flow is from the upper left to lower right. The floodplain area is delimited by a 10 y flood (3000 m^3/s)collected in the spring (38 sites), summer (24 sites), and winter (34 sites) of 2004.

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Every effort was made to collect samples from all water bodies in the study area. Different physico-chemical variables were then analyzed. To determine differences caused by different hydrological conditions data sets were individually subjected to multivariate statistical analyses involving Principal Component Analysis (PCA) and Cluster analysis based on a Ward algorithm (SPSS© 14.0 package). PCA

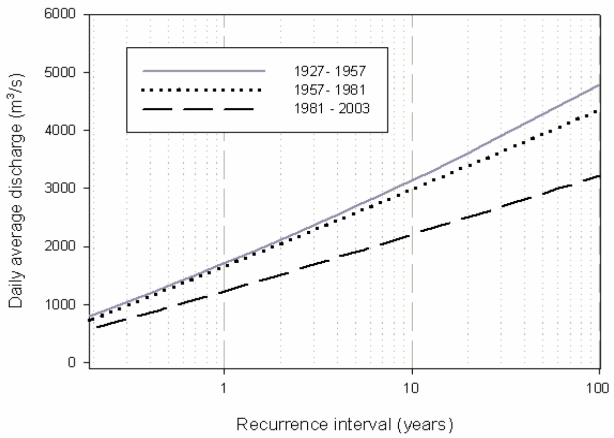

Figure 3. Magnitude-frequency plots illustrating reduced frequency of high-discharge events in more recent years.

Period	Peak Discharge (m3/s)	Events		Duration (d)		Cumulative Discharge (m3/s)	
		total	per year	total	per event	total	per event
1927-1957	600 < x > 2100	206	7.10	811	3.94	738098.95	3583.00
	x > 2100	21	0.72	335	15.95	437880.21	20851.44
	TOTAL	**227**	**7.82**	**1146**		**1175979.16**	
1957-1981	600 < x > 2025	146	5.84	669	4.58	593994.61	4068.46
	x > 2025	20	0.80	310	15.50	406952.18	20347.61
	TOTAL	**166**	**6.64**	**979**		**1000946.79**	
1981-2003	600 < x > 1465	90	4.09	303	3.37	246465.12	2738.50
	x > 1465	16	0.73	161	10.06	180496.64	11281.04
	TOTAL	**106**	**4.82**	**464**		**426961.76**	

Table 1. Number, duration and accumulated mean discharge of flood events at the Zaragoza gauging station. Data were analyzed separately for the three temporal periods. Flood events were categorized according to the estimated bankfull discharge.

factors and the physico-chemical features of cluster groups were used to identify the main hydrogeochemical processes operating at the study reach. Afterwards, wetlands types were defined using the cluster solutions for the three examined periods. To facilitate the identification of wetland types, the potential influence of alluvial aquifer was evaluated using total dissolved solids (TDS) and nitrate concentrations. To interpret the results under a reach perspective, a geomorphological characterization of the riverscape (sensu Malard et al., 200) was performed.

In the **study III,** The hydroperiod, flooded sediments and riparian understory were examined for eight riparian wetlands. Hydroperiod measurements were performed comparing water level variation at each wetland with that on the Ebro River. To accomplish that, continuous water-level height meters were places at each wetland, and levels were measured during one ordinary flood (536 m^3/s; 0.15 y return period) that peaked on April 24, 2006. Secondly, we chose sediment physico-chemical variables to relate them with wetland hydroperiod. Sampling was performed at flooded areas. Two different seasons with contrasting river discharges were selected for collecting sediment samples (0-3 cm) in the upstream, central, and downstream section of the flooded zone. Then, sediment physico-chemical variables were analyzed. Differences between high and low water levels were determined by individually subjecting winter and summer data to multivariate statistical analysis that incorporated Principal Components Analysis (PCA) with the Factor analysis procedure, and Cluster Analysis based on a Ward algorithm (SPSS© 14.0 package). The multivariate analysis was used to identify factors that accounted for data variability as well as to identify wetland types. Finally, three transects were placed at the upstream, central, and downstream sections of the riparian wetland sites in August 2005. Sampling plots were set every 5 m along transects .For each plot, the ground cover for each understory species was visually estimated within a 1-m^2 quadrant. Then, Cumulative Diversity (CD) spectra were drawn to explore the effects of geomorphological features on the riparian understory structure. Additional characterizations of wetland successional stage were provided using different diversity measures (see study III for details) of the riparian understory.

In the study **IV**, soil organic carbon and nitrogen concentrations of riparian patches in the Middle Ebro River Floodplain that differed in landform development were examined, and sediment inputs from the river were characterized. The data from 1956 and 1998 aerial pictures were used to create a transition map. Four natural and two artificial landform categories were set using the transition map. Three natural riparian zones were chosen because they encompassed all sampling categories. At each study plot, topsoil samples (10 cm) were collected. Organic and refractory Carbon, as well as Nitrogen was analyzed for soil samples. A one-way ANOVA test was then performed using SPSS® 14.0 to test for significant differences between categories in each of the riparian areas. An ANCOVA test was performed to check the influence of landform evolution on the organic matter at the substrate on natural patches. Secondly, sediment traps, made of artificial grass mats, were used to collect the sediment deposited by a single flood on 1 January 2006, which reached 754,44 m^3/s at the Zaragoza gauge station (12 km upstream from the study area). The relationships between the sediment grain size and the quantity of carbon and nitrogen deposited in the examined flood were combined with soil sample results in order to estimate the incorporation of in-situ produced organic matter to the riparian substrates.

In the **study V**, soil profiles of four riparian habitats in one reach of the Middle Ebro River (NE Spain) were examined. To investigate the spatial and temporal heterogeneity on accretion patterns, organic and refractory carbon, nitrogen and grain size were examined for the collected profiles. C:N ratios were also calculated in order to obtain a rough indicator of the organic matter quality. Then, sediment, organic and refractory carbon and nitrogen accretion rates during the last century were estimated using the Cs-137 dating and information from aerial photographs. To test the theoretical influence of grain size over organic matter accretion, the relationship between organic and refractory carbon, nitrogen and C:N with the fines (<63 μm) fraction was examined. To analyze the effect of organic matter accretion patterns over organic matter biochemical complexity, the relationship between refractory carbon and C:N with an organic incorporation index (see study V for details) was also examined. Finally, periods with different carbon and nitrogen accretion patterns were identified for the examined habitats.

4. Results and discussion

4.1 Human disruption of the River disturbance regime

The river disturbance regime is the main factor explaining floodplain structure and organic matter accretion patterns in the Middle Ebro Floodplains. Consequently, human alterations of this primary driver are reflected on these ecosystems. River-floodplain interactions have been disrupted at different spatio-temporal scales. Geomorphological dynamics no longer take place in the study reach (Fig.1). The absence of channel migration since 1981 has probably caused vertical accretion to dominate floodplain formation. As a consequence, autogenous processes dominate riparian succession at the majority of the study reach. It is reflected on landscape and riparian understory diversity (**study I, III**), as well as on the potential of these ecosystems as organic matter sinks (**study IV, V**). With regards to hydrological connectivity, the majority of the floodplain area is flooded only during high magnitude floods (> 2500 m^3/s, 5.36 y return period) and remains disconnected from the river most of the time. River-floodplain interactions are restricted to small areas close to the main channel ("straitjacket" area *sensu* Lamers et al., 2006) during short periods. It shortens, both in space and time, the current hydroperiod, hydrogeochemical and flooded sediment heterogeneity (**study II, III**).

4.2 Landscape dynamics during the last century

Landscape structure and dynamic adjusted to changes in the flow regime, flood protection structures and occupation of the river space (**study I**). The highest ecotope diversity was observed prior to the 1960´s because it was during that time that construction of major flow regulation infrastructures began. Flood events (Tab.1) maintained a diverse array of landforms in areas adjacent to the main channel at that period. Mature stages were dispatched to the outer floodplain, although agricultural expansion in the floodplain between 1946 and 1956 substantially lowered the heterogeneity at those areas. After 1956, the drop in ecotope diversity can be explained in part by changes in hydrology, which promoted the decrease of ecotopes at the initial successional stages. However, the magnitude of river flow and fluctuations in river discharge between 1957 and 1981 did not greatly differ from the previous period (Fig.3). Therefore, it was the extensive implementation of flood protection structures along the river banks that was the most likely culprit in disrupting river-floodplain interactions. As a consequence, succession was probably accelerated by higher vertical accretion rates once main channel adjusted to dykes. After 1981, the progressive dominance of mature ecotopes since then, what results in the lowest ecotope diversity, has been in response to the synergic effect of flow regulation and flood protection. Ecotope rejuvenation nearly disappeared whereas succession dominates landscape dynamics. As an example, only bank erosion at localized points was detected between 1998 and 2003, despite a large flood (February 2003) took place in this time span.

4.3 Riparian understory diversity as indicator of riverscape heterogeneity

The structure of the riparian understory was valid to asses the specific wetland successional stage of the examined sites (**study III**). The shape of the cumulative diversity spectra (estimated for the upstream, central and downstream areas of the examined wetlands) mirrors the local wetland topography, what helps to interpret the wetland successional stage. The planar forms in central sections of the spectra indicate the dominance of highly connected plots, with the absence of vegetation or dominance by a single macrophyte species. Discrete or continuous environmental gradients are reflected in the rising phases of the spectra; diagonal forms indicate sequential gradients, characteristic of early successional stages. Asymptotic forms correspond to step-wise gradients, characteristic of mature stages. At the selected wetlands, asymptotic shapes predominated due to the dominance of vertical accretion in the floodplain formation process. In some cases, natural trends are completely overridden by the human occupation of areas surrounding water bodies. With regards to the riverscape (array of riparian wetlands, see Malard et al., 2000) heterogeneity, a two-dimensional conceptual model was built using the information provided by our diversity measures. Low species turnover rates lead to homogenization and low Cumulative Diversity Rate (CDR) values, while low Cumulative Diversity (CD) values result if environmental conditions are such that they prevent species from establishing along the section (see study III for CD and CDR calculations). Most of the wetlands examined were classified in our model within categories with low environmental heterogeneity, what stresses the fact that the riverscape at the study reach is homogenous and dominated by mature stages with step-wise environmental gradients.

4.4 Hydrological connectivity

The effect of mass and energy exchange between the floodplain and the main channel during floods, i.e. hydrological connectivity, explained the hydrogeochemical dynamics of surface waters at the study reach, as well as the hydroperiod and sediment heterogeneity for one group of eight riparian wetlands (study II, III). At reach scale, between the water variables associated with mineralization (TDS, SO_4^-, Cl^-, Na^+, Ca^{++}, Mg^{++}, K^+), and those associated with fertilization (NO_3^-, NH_4^+, PO_4^-, N-org) as the first two major components of the PCA indicate that dilution and dissolution, which are associated with river and alluvial aquifer water discharges, and biogeochemical processes associated with water pathways are the major factors influencing the characteristics of the wetlands at the study reach. Others have noted that this balance is the main factor in determining the shifting physico-chemical mosaic of the riverscape during flood events of varying magnitudes (Malard et al., 2000; Arntzen et al., 2006; Malard et al., 2006). During floods, main channel inputs of low mineralized and turbid waters differentiated the water chemistry of superficially connected wetlands. Conversely, lower suspended solids content and more saline waters characterized less superficially connected wetlands during floods, possibly reflecting the relative dominance of groundwater inputs. Our results suggest that the balance of river seepage and inputs from a saline alluvial aquifer explain the spatial and temporal heterogeneity in water mineralization at these groundwater-fed wetlands. With regards to the nutrients, our results confirm previous reports of the importance of nutrient inputs from the main river channel to adjacent floodplain during flood-pulses (Van der Brink et al., 1993; Heiler et al., 1995; Hein et al., 1999; Tockner et al., 1999). With respect to the sub-surface connected wetlands, the spatial and temporal heterogeneity of biogeochemical processes at the interface of riparian and aquatic ecosystems (Mitsch & Gosselink, 1993; Haycock et al., 1996; Dahm et al., 1998; McClain et al., 2003; Weng et al., 2003) were thought to influence nutrient concentrations. On the basis of surface water chemistry during all examined floods three types of wetlands were identified and associated with the dominant water sources during different magnitude flood.

At site scale, four types of wetland hydroperiod were identified during an ordinary flood (536 m^3/s, 0.15 y return period) using the hysteretic loop method (see section III for details): 1) Backflow channels permanently connected at their downstream end; 2) Sites where the hydroperiod was controlled by superficial inputs at a specific river-discharge threshold during the rising phase of the flood; 3) Groundwater-connected sites where rapid fluctuations in water level responded to variations at the hyporheic zone; 4) Superficially disconnected oxbow lakes which had a thick layer of fine sediment that impeded vertical connectivity. With regards to sediment structure (0-3 cm), the results of the applied multivariate approach seemed consistent with the hysteretic loop evaluations. The sediments of the disconnected wetlands (Hydroperiod type 4) were found to be more organic due to the incorporation of in-situ produced organic matter to the substrate. Conversely, the majority of the sediments at connected wetlands (Hydroperiod types 1, 2 and 3) were found to be more inorganic, since the sedimentation of river material and the exchange of organic matter with the main channel, decreased the organic content of the sediment. Seasonal variability in river discharge was found to have functional implications for substrates at the study locations depending on wetland hydroperiod. Recurrent floods prior to winter sampling created a gradient from connected sites, which had diluted environments with low ammonia/nitrate ratios, to disconnected sites, where conductivity and organic matter were elevated and reduced nitrogen forms predominated. In summer, stagnant conditions caused conductivity to rise relative to values in winter.

According to the previous results, we can conclude that hydrogeochemical, hydroperiod and sediment heterogeneity described for the low-magnitude floods examined (<750 m^3/s flood, 0.22 y) is relatively high. However, it is not consistent if larger spatio-temporal scales are considered. The observed heterogeneity was mostly limited to permanently flooded areas, which accounts for about 0.7 % of the total floodplain area. Moreover, those areas only covered the 30 % of natural riparian wetlands. In addition, floods may increase the similarity among aquatic habitats in our study area. Connected wetlands accounted for a very small portion of the floodplain and were highly accreted suggesting that potentially flooded areas are highly terrestrialized in the study reach. We hypothesize that water sources will rapidly become dominated by superficial inputs during events larger than those considered in this study. In turn, the area covered by disconnected wetlands is relatively bigger and dominates over other wetland types, although the portion of permanently flooded area is also low. At those sites, groundwater connectivity is counteracted by thick layers of bottom sediment, whereas they remain superficially disconnected most of the time. With regards to the sediment, functionality of submerged substrates is restricted to small areas during short periods.

4.5 Organic matter accretion at floodplain substrates

C and N accretion patterns at the study reach have been severely modified during the last century (study V). Anthropic influences have lowered the importance of allochthonous OM inputs. It reduced the potential of the Middle Ebro floodplains as C and N sinks because this function maximizes when fine sediment deposition (< 2 mm) is higher. At connected ecotopes, sedimentation is two-fold smaller if compared with early successional stages of currently disconnected habitats. At the disconnected ecotopes, the most recently deposited sediments (h= 30 cm) does not seem to be affected by inputs of inorganic river seston, whereas carbon and nitrogen budgets are markedly higher after 1963. It highlights the inverse relationship between external inputs and organic matter incorporation at floodplain substrates (Rostan et al., 1987; Schwarz et al., 1996; Tockner and Schiemer, 1997; Daniels, 2003; Mitsch et al. 2005). Such historical evolution on the accretion patterns support the use of landform geomorphological evolution for a rapid assessment of the organic matter budgets at surface sediments (0-10 cm). Geomorphological dynamics can indicate the current predominance of autochthonous vs. allochthonous organic matter inputs for a given ecotope, what influences the organic carbon and nitrogen stocks at the top-soil (study IV). The stocks of carbon and nitrogen were highest in the topsoil of the substrates underlying the mature stages of vegetation particularly *Populus alba*. The incorporation of organic carbon and nitrogen produced in situ was evident in the topsoil of those old patches (>60 y) In turn, stocks were the lowest in the younger patches (<60 y), more hydrologically connected, where the incorporation of in-situ produced organic matter was impeded by higher sedimentations rates, as well as by the export of inorganic sediments and organic matter during floods. Finally, the amounts of carbon and nitrogen in the topsoil of patches used for intensive agriculture and poplar production were similar to those in young natural patched, even though that those patches are old. Agricultural fields and poplar groves are usually protected against floods, so suspended sediments are not deposited in these patches, while primary production is not entirely logged on soils.

With regards to the quality, the shift towards the dominance of organic matter inputs did not promote any significant change in the size of the refractory organic matter pool, although influenced data variability. At more connected stages, particulate and sediment-associated organic matter quality is deposited over the surface during floods. Due to the variability at origin, allochthonous organic matter inputs might be more variable if compared with in-situ produced organic matter, which is though to be more homogenous in terms of quality. It was confirmed by the analysis of top-soil samples. Consequently, our results indicate that the size of the refractory SOM pool relied on the quantity of organic carbon rather than on its biochemical complexity. About 50% of the organic matter accreted at the middle Ebro floodplains can be considered in the passive pool, although the role of physical protection must be further analyzed prior to establish the size of the protected pool. Although the system functioning had been so heavily modified, the middle Ebro floodplains have a high capacity to accumulate C and N, especially when hydrological connectivity with the main channel is high enough to ensure the dominance of external inputs. C and N accretion rates were in the highest range for those reported for another wetlands worldwide, exceeding those ranges when allochthonous inputs dominated. Even at lower current levels, the potential is more than an order of magnitude greater than has been reported for peatlands or afforestation of agricultural fields.

4.6 Ecological status of the study reach: Implications for restoration and management

According to our results, it seems clear that the effects of diminished river-floodplain interactions have clearly affected the ecological status of the study reach. To restore these ecosystems, rehabilitation efforts should be paid at different scales. As inferred from the landscape analysis, both the natural flow regime and floodplain topography should be subjected for and effective restoration. Landscape patterns will limit the restoration success if the natural flow regime is recovered to a more historical condition. Although surface and groundwater connectivity would be enhanced, the erosive effect of floods would be strongly counteracted by artificial dykes, at convex banks, and vegetation encroachment, at concave banks. Similarly, hydrological patterns will limit the restoration success if floodplain topography is modified to enhance connectivity. A potentially dynamic corridor could be created if dykes are removed and floodplain height is lowered, but the current flow regime is not adequate to ensure self-sustained processes over the entire area.

New management strategies should therefore be implemented at basin, reach and site scale. At basin scale, alternative strategies would need to make more integrated use of natural resources, primarily soil and water, as suggested by Comín (1999). Such management strategies should ensure erosive floods (> 3000 m^3/s) at least to a certain extent, and also a range of seasonal discharge fluctuations below bankfull (1600 m^3/s, 1927-2003). Nevertheless, it is unrealistic to attempt to restore the magnitude of river discharge under the current basin management program. Instead, efforts should focus on mimicking the other components of the flow regime since that will more likely produce positive results that will contribute to sustainable floodplain dynamics (Poff et al., 1997; Bendix and Hupp, 2000; Hughes and Rood, 2003; Stromberg et al., 2007). At reach scale, it is necessary to remove or redirect dikes and lowering floodplain heights to restore an ecosystem with the hydrogeomorphological capacity to maintain the diverse mosaic of both riparian wetlands and forests which is characteristic of a well-preserved floodplain. To maximize the accumulation of organic matter, the conversion of agricultural fields and poplar groves into natural land-covers habitats is required. By increasing allochthonous inputs, carbon and nitrogen accretion rates would be maximized. It would also match another restoration targets as increasing landscape diversity or water quality enhancement. At site scale, hydroperiod diversity could be increased by creating artificial wetlands or modifying the local topography of existing wetlands. At the end of study III, recommendations for wetlands included in this study are proposed. It would enlarge the extent of flooded area during different magnitude events, therefore increasing heterogeneity on hydrogeochemical dynamics and sediment structure.

To perform all this restoration strategies, periodic inputs of energy, through significant economic investments, will be required. Self-sustained restoration, as would be desired, will no longer take place under the current basin policies. It has been suggested for other european rivers (Schiemer et al., 1999; Hughes et al., 2001, Buijse et al., 2002; Baptist et al., 2004) as a valid compromise between the need for flood protection and the desire for ecosystem rehabilitation in highly regulated rivers. Moreover, we strongly believe that this is an investment rather than a cost, since multiple benefits will be provided by floodplains once restored. It is also highly appropriated to carry out additional monitoring of restoration if performed, to evaluate the success of previously-defined restoration plans, redefining them if key parameters change (Hughes et al., 2005).

5. References

Adams, W. M., and A. Perrow. 1999. Scientific and institutional constraints on the restoration of European floodplains. . p. 89-97. In S. B. Marriot, & J. Alexander (eds.), Floodplains: Interdisciplinary Approaches. Geological Society, London.

Amoros, C. 2001. The concept of habitat diversity between and within ecosystems applied to river side-arm restoration. Environmental Management 28: 805-817.

Amoros, C., and G. Bornette. 1999. Antagonistic and cumulative effects of connectivity: a predictive model based on aquatic vegetation in riverine wetlands. Archiv Fur Hydrobiologie: 311-327.

Amoros, C., and G. Bornette. 2002. Connectivity and biocomplexity in waterbodies of riverine floodplains. Freshwater Biology 47: 761-776.

Amoros, C., and G. E. Petts. 1993. Hydrosystèmes Fluviaux. Masson Paris.

Amoros, C., and P. M. Wade. 1996. Ecological succesions. p. 211-241. In G. E. Petts, & C. Amoros (eds.), Fluvial Hydrosystems. Chapman & Hall, London.

Arntzen, E. V., D. R. Geist, and P. E. Dresel. 2006. Effects of fluctuating river flow on groundwater/surface water mixing in the hyporheic zone of a regulated, large cobble bed river. River Research and Applications 22: 937-946.

Asselman, N. E. M., and H. Middelkoop. 1995. Floodplain Sedimentation - Quantities, Patterns and Processes. Earth Surface Processes and Landforms 20: 481-499.

Baker, M. A., and P. Vervier. 2004. Hydrological variability, organic matter supply and denitrification in the Garonne River ecosystem. Freshwater Biology 49: 181-190.

Baptist, M. J., W. E. Penning, H. Duel, A. J. M. Smits, G. W. Geerling, G. E. M. van der Lee, and J. S. L. Van Alphen. 2004. Assessment of the effects of cyclic floodplain rejuvenation on flood levels and biodiversity along the Rhine river. River Research and Applications 20: 285-297.

Batalla, R. J., C. M. Gomez, and G. M. Kondolf. 2004. Reservoir-induced hydrological changes in the Ebro River basin (NE Spain). Journal of Hydrology 290: 117-136.

Begueria, S., J. I. Lopez-Moreno, A. Gomez-Villar, V. Rubio, N. Lana-Renault, and J. M. Garcia-Ruiz. 2006. Fluvial adjustments to soil erosion and plant cover changes in the Central Spanish Pyrenees. Geografiska Annaler Series a-Physical Geography 88A: 177-186.

Begueria, S., J. I. Lopez-Moreno, A. Lorente, M. Seeger, and J. M. Garcia-Ruiz. 2003. Assessing the effect of climate oscillations and land-use changes on streamflow in the Central Spanish Pyrenees. Ambio 32: 283-286.

Bendix, J., and C. R. Hupp. 2000. Hydrological and geomorphological impacts on riparian plant communities. Hydrological Processes 14: 2977-2990.

Bornette, G., C. Amoros, and N. L. Lamouroux. 1998. Aquatic plant diversity in riverine wetlands: The role of connectivity. Freshwater Biology 39: 267-283.

Brookes, A., J. Baker, and C. Redmond. 1996. Floodplain restoration and riparian zone management. p. 201-229. In A. Brookes, & F. D. Shields (eds.), River channel restoration: Guiding principles for Sustainable Projects. John Wiley and Sons, Chichester.

Brunke, M. 2002. Floodplains of a regulated southern alpine river (Brenno, Switzerland): ecological assessment and conservation options. Aquatic Conservation: Marine and Freshwater Ecosystems 12: 583-599.

Buijse, A. D., H. Coops, M. Staras, L. H. Jans, G. J. Van Geest, R. E. Grift, B. W. Ibelings, W. Oosterberg, and F. C. J. M. Roozen. 2002. Restoration strategies for river floodplains along large lowland rivers in Europe. Freshwater Biology 47: 889-907.

Castro, P., J. Guerrero, and M. A. Muñoz. 2001. Plan de Restauración del Bosque de Ribera en la Reserva Natural de los Galachos (Zaragoza). Consejo de Proteccion de la Naturaleza de Aragon, Zaragoza.

Comin, F. A. 1999. Management of the Ebro River Basin: Past, present and future. Water Science and Technology 40: 161-168.

Constanza, R., R. d´Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. V. Neill, J. Paruelo, R. G. Raskin, P. Sutton, and M. Van der Belt. 1997. The value of the world´s ecosystem services and natural capital. Nature 387: 253-260.

Dahm, C. N., N. B. Grimm, P. Marmonier, H. M. Valett, and P. Vervier. 1998. Nutrient dynamics at the interface between surface waters and groundwaters. Freshwater Biology 40: 427-451.

Daniels, J. M. 2003. Floodplain aggradation and pedogenesis in a semiarid environment. Geomorphology 56: 225-242.

Day, J. W., J. Y. Ko, J. Rybczyk, D. Sabins, R. Bean, G. Berthelot, C. Brantley, L. Cardoch, W. Conner, J. N. Day, A. J. Englande, S. Feagley, E. Hyfield, R. Lane, J. Lindsey, J. Mistich, E. Reyes, and R. Twilley. 2004. The use of wetlands in the Mississippi Delta for wastewater assimilation: a review. Ocean & Coastal Management 47: 671-691.

de Domitrovic, Y. Z. 2003. Effect of fluctuations in water level on phytoplankton development in three lakes of the Parana river floodplain (Argentina). Hydrobiologia 510: 175-193.

Dynesius, M., and C. Nilsson. 1994. Fragmentation and Flow Regulation of River Systems in the Northern 3rd of the World. Science 266: 753-762.

Frutos, L. M., A. Ollero, and M. Sánchez Fabre. 2004. Caracterización del Ebro y su cuenca y variaciones en su comportamiento hidrológico. p. 223-280. In A. Gil Oncina (ed.), Alteración de los regimenes fluviales peninsulares. Fundación Caja Murcia, Alicante.

Galat, D. L., L. H. Fredrickson, D. D. Humburg, K. J. Bataille, J. R. Bodie, J. Dohrenwend, G. T. Gelwicks, J. E. Havel, D. L. Helmers, J. B. Hooker, J. R. Jones, M. F. Knowlton, J. Kubisiak, J. Mazourek, A. C. McColpin, R. B. Renken, and R. D. Semlitsch. 1998. Flooding to restore connectivity of regulated, large-river wetlands - Natural and controlled flooding as complementary processes along the lower Missouri River. Bioscience 48: 721-733.

Garcia-ruiz, J. M., T. Lasanta, L. Ortigosa, P. Ruizflano, C. Marti, and C. Gonzalez. 1995. Sediment yield under different land uses in the spanish pyrenees. . Mountain Research and Development 15: 229-240.

Geerling, G. W., A. M. J. Ragas, R. Leuven, J. H. van den Berg, M. Breedveld, D. Liefhebber, and A. J. M. Smits. 2006. Succession and rejuvenation in floodplains along the river Allier (France). Hydrobiologia 565: 71-86.

Haycock, H., T. Burt, K. Goulding, and G. Pinay. 1996. Buffer zones: Thier Processes and Potential in Water Protection. Qest Environmental, Harpenden, Hertfordshire.

Hefting, M., J. C. Clement, D. Dowrick, A. C. Cosandey, S. Bernal, C. Cimpian, A. Tatur, T. P. Burt, and G. Pinay. 2004. Water table elevation controls on soil nitrogen cycling in riparian wetlands along a European climatic gradient. Biogeochemistry 67: 113-134.

Heiler, G., T. Hein, F. Schiemer, and G. Bornette. 1995. Hydrological connectivity and flood pulses as the central aspects for the integrity of a river-floodplain system. Regulated Rivers-Research & Management 11: 351-361.

Hein, T., C. Baranyi, G. Heiler, C. Holarek, P. Riedler, and F. Schiemer. 1999. Hydrology as a major factor determining plankton development in two floodplain segments and the River Danube, Austria. Archiv Fur Hydrobiologie: 439-452.

Hein, T., C. Baranyi, G. J. Herndl, W. Wanek, and F. Schiemer. 2003. Allochthonous and autochthonous particulate organic matter in floodplains of the River Danube: the importance of hydrological connectivity. Freshwater Biology 48: 220-232.

Hein, T., C. Baranyi, W. Reckendorfer, and F. Schiemer. 2004. The impact of surface water exchange on the nutrient and particle dynamics in side-arms along the River Danube, Austria. Science of the Total Environment 328: 207-218.

Hein, T., G. Heiler, D. Pennetzdorfer, P. Riedler, M. Schagerl, and F. Schiemer. 1999. The Danube restoration project: Functional aspects and planktonic productivity in the floodplain system. Regulated Rivers-Research & Management 15: 259-270.

Henry, C. P., and C. Amoros. 1995. Restoration Ecology of Riverine Wetlands .1. A Scientific Base. Environmental Management 19: 891-902.

Hohensinner, S., H. Habersack, M. Jungwirth, and G. Zauner. 2004. Reconstruction of the characteristics of a natural alluvial river-floodplain system and hydromorphological changes following human modifications: The Danube River (1812-1991). River Research and Applications 20: 25-41.

Hontoria, C., J. C. Rodriguez-Murillo, and A. Saa. 1999. Relationships between soil organic carbon and site characteristics in peninsular Spain. Soil Science Society of America Journal 63: 614-621.

Hughes, F. M. R., W. M. Adams, E. Muller, C. Nilsson, K. S. Richards, N. Barsoum, H. Decamps, R. Foussadier, J. Girel, H. Guilloy, A. Hayes, M. Johansson, L. Lambs, G. Pautou, J. L. Peiry, M. Perrow, F. Vautier, and M. Winfield. 2001. The importance of different scale processes for the restoration of floodplain woodlands. Regulated Rivers-Research & Management 17: 325-345.

Hughes, F. M. R., A. Colston, and J. O. Mountford. 2005. Restoring riparian ecosystems: The challenge of accommodating variability and designing restoration trajectories. Ecology and Society 10: 12.

Hughes, F. M. R., and S. B. Rood. 2003. Allocation of river flows for restoration of floodplain forest ecosystems: A review of approaches and their applicability in Europe. Environmental Management 32: 12-33.

Ibanez, C., N. Prat, and A. Canicio. 1996. Changes in the hydrology and sediment transport produced by large dams on the lower Ebro river and its estuary. Regulated Rivers-Research & Management 12: 51-62.

IPCC. 2000. Land Use, Land Use Change, and Forestry. Cambridge University Press, Cambridge. United Kingdom.

Johnston, C. A. 1991. Sediment and nutrient retention by freshwater wetlands- Effects on surface-water quality. Critical Reviews in Environmental Control 21: 491-565.

Junk, W. J., B. Bayley, and R. E. Sparks. 1989. The flood pulse concept in river-floodplain systems. in D.P.Dodge (ed) Proc. Intern. Large River Symposium. Special Issue of J.Canad. Fish and Aquat. Sci. 106: 11-27.

Knowlton, M. F., and J. R. Jones. 1997. Trophic status of Missouri River floodplain lakes in relation to basin type and connectivity. Wetlands 17: 468-475.

Lamers, L. P. M., R. Loeb, A. M. Antheunisse, M. Miletto, E. C. H. E. T. Lucassen, A. W. Boxman, A. J. P. Smolders, and J. G. M. Roelofs. 2006. Biogeochemical constraints on the ecological rehabilitation of wetland vegetation in river floodplains. Hydrobiologia 565: 165-186.

Langhans, S. D., and K. Tockner. 2006. The role of timing, duration, and frequency of inundation in controlling leaf litter decomposition in a river-floodplain ecosystem (Tagliamento, northeastern Italy). Oecologia 147: 501-509.

Lockaby, B. G., R. Governo, E. Schilling, G. Cavalcanti, and C. Hartsfield. 2005. Effects of sedimentation on soil nutrient dynamics in riparian forests. Journal of Environmental Quality 34: 390-396.

Lockwood, L. J., and S. L. Pimm. 1999. When does restoration succeed ? p. 363-392. In E. Weiher, & P. A. Keddy (eds.), Ecological

Assembly Rules: Perspectives, Advances, Retreats. Cambidge University Press, Cambridge.

Lopez-Moreno, J. I., S. Begueria, and J. M. Garcia-Ruiz. 2006. Trends in high flows in the central Spanish Pyrenees: response to climatic factors or to land-use change? Hydrological Sciences Journal-Journal Des Sciences Hydrologiques 51: 1039-1050.

Malard, F., K. Tockner, and J. V. Ward. 2000. Physico-chemical heterogeneity in a glacial riverscape. Landscape Ecology 15: 679-695.

Malard, F., U. Uehlinger, R. Zah, and K. Tockner. 2006. Flood-pulse and riverscape dynamics in a braided glacial river. Ecology 87: 704-716.

McClain, M. E., E. W. Boyer, C. L. Dent, S. E. Gergel, N. B. Grimm, P. M. Groffman, S. C. Hart, J. W. Harvey, C. A. Johnston, E. Mayorga, W. H. McDowell, and G. Pinay. 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6: 301-312.

McGarigal, K., and B. Marks. 1995. FRAGSTATS: Spatial Analysis Program for Quantifiying Landscape Structure. USDA For. Serv. Gen. Tech. Rep. PNW-351.

Megonigal, J. P., W. H. Conner, S. Kroeger, and R. R. Sharitz. 1997. Aboveground production in Southeastern floodplain forests: A test of the subsidy-stress hypothesis. Ecology 78: 370-384.

Metzger, J. P., and H. Décamps. 1997. The structural connectivity treshold : an hypothesis in conservation biology at landscape scale. Acta Oecologica 18: 1-12.

Mitsch, W. J., and J. G. Gosselink. 2000. Wetlands. Van Nostrand Reinhold, New York.

Mitsch, W. J., L. Zhang, C. J. Anderson, A. E. Altor, and M. E. Hernandez. 2005. Creating riverine wetlands: Ecological succession, nutrient retention, and pulsing effects. Ecological Engineering 25: 510-527.

Naiman, R. J., S. E. Bunn, C. Nilsson, G. E. Petts, G. Pinay, and L. C. Thompson. 2002. Legitimizing fluvial ecosystems as users of water: An overview. Environmental Management 30: 455-467.

Naiman, R. J., and H. Decamps. 1997. The ecology of interfaces: Riparian zones. Annual Review of Ecology and Systematics 28: 621-658.

Naiman, R. J., J. J. Magnunson, D. M. McKnight, and J. A. Stanford. 1995. The Freshwater Imperative - A research Agenda. Island Press, Alstone, Washington, D.C.

Noe, G. B., and C. R. Hupp. 2005. Carbon, nitrogen, and phosphorus accumulation in floodplains of Atlantic Coastal Plain rivers, USA. Ecological Applications 15: 1178-1190.

Ollero, A. 1992. Los meandros libres de el Ebro (Logroño-La Zaida). Geomorfología fluvial, ecogeografía y Riesgos. Tesis Doctoral, Zaragoza, Zaragoza.

Ollero, A. 1995. Dinámica reciente del cauce de el Ebro en la Reserva Natural de los Galachos. Cuaternario y Geomorfología 9: 85-93.

Ollero, A. 2007. Channel adjustments, floodplain changes and riparian ecosystems of the middle Ebro River: Assessment and management. International Journal of Water Resources Development 23: 73-90.

Olson, D. M., and E. Dinerstein. 1998. The global 200: A representation approach to conserving the Earth's most biologically valuable ecoregions. Conservation Biology 12: 502-515.

Paul, E. A., S. J. Morris, R. T. Conant, and A. F. Plante. 2006. Does the acid hydrolysis-incubation method measure meaningful soil organic carbon pools? Soil Science Society of America Journal 70: 1023-1035.

Petts, G. E., H. Moller, and A. L. Roux. 1989. Historical Change of Large Alluvial Rivers: Western Europe. Jhon Wiley and Sons, Chichester.

Pinay, G., J. C. Clement, and R. J. Naiman. 2002. Basic principles and ecological consequences of changing water regimes on nitrogen cycling in fluvial systems. Environmental Management 30: 481-491.

Pinay, G., A. Fabre, P. Vervier, and F. Gazelle. 1992. Control of C,N,P distribution in soils of riparian forest. Landscape Ecology 6: 121-132.

Pinilla, V. 2006. The development of irrigated agriculture in twentieth-century Spain: a case study of the Ebro basin. Agricultural History Review 54: 122-141.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime. Bioscience 47: 769-784.

Post, W. M., and K. C. Kwon. 2000. Soil carbon sequestration and land-use change: processes and potential. Global Change Biology 6: 317-327.

Ravenga, C., S. Murray, J. Abramovitz, and A. Hammond. 1998. Watersheds of the World. World Resources Institute, Washington, DC. USA.

Regato, P. 1988. Contribucion al estudio de la flora y la vegetación del galacho de la Alfranca en relación con la evolución del sistema fluvial. Diputación General de Aragon, Zaragoza.

Robertson, A. I., P. Bacon, and G. Heagney. 2001. The responses of floodplain primary production to flood frequency and timing. Journal of Applied Ecology 38: 126-136.

Robertson, A. I., S. E. Bunn, P. I. Boon, and K. F. Walker. 1999. Sources, sinks and transformations of organic carbon in

Australian floodplain rivers. Marine and Freshwater Research 50: 813-829.

Rostan, J. C., C. Amoros, and J. Juget. 1987. The Organic Content of the Surficial Sediment - a Method for the Study of Ecosystems Development in Abandoned River Channels. Hydrobiologia 148: 45-62.

Rovira, P., C. Kurz-Besson, M. M. Couteaux, and V. R. Vallejo. 2008. Changes in litter properties during decomposition: A study by differential thermogravimetry and scanning calorimetry. Soil Biology & Biochemistry 40: 172-185.

Schiemer, F. 1995. Restoration of floodplains-possibilities and constraints. Archive für Hydrobiologie 101: 383-398.

Schiemer, F., C. Baumgartner, and K. Tockner. 1999. Restoration of floodplain rivers: The 'Danube restoration project'. Regulated Rivers-Research & Management 15: 231-244.

Schwarz, W. L., G. P. Malanson, and F. H. Weirich. 1996. Effect of landscape position on the sediment chemistry of abandoned-channel wetlands. Landscape Ecology 11: 27-38.

Simons, J., C. Bakker, M. H. I. Schropp, L. H. Jans, F. R. Kok, and R. E. Grift. 2001. Man-made secondary channels along the River Rhine (the Netherlands); results of post-project monitoring. Regulated Rivers-Research & Management 17: 473-491.

Sparks, R. E. 1995. Need for Ecosystem Management of Large Rivers and Their Floodplains. Bioscience 45: 168-182.

Stanford, J. A., M. S. Lorang, and F. R. Hauer. 2005. The shifting habitat mosaic of river ecosystems. Verhandlungen International Verein. Limnologie 29: 123-136.

Stanford, J. A., J. V. Ward, W. J. Liss, C. A. Frissell, R. N. Williams, J. A. Lichatowich, and C. C. Coutant. 1996. A general protocol for restoration of regulated rivers. Regulated Rivers-Research & Management 12: 391-413.

Steiger, J., and A. M. Gurnell. 2003. Spatial hydrogeomorphological influences on sediment and nutrient deposition in riparian zones: observations from the Garonne River, France. Geomorphology 49: 1-23.

Stromberg, J. C., V. B. Beauchamp, M. D. Dixon, S. J. Lite, and C. Paradzick. 2007. Importance of low-flow and high-flow characteristics to restoration of riparian vegetation along rivers in and south-western United States. Freshwater Biology 52: 651-679.

Tabacchi, E., D. L. Correll, R. Hauer, G. Pinay, A. M. Planty-Tabacchi, and R. C. Wissmar. 1998. Development, maintenance and role of riparian vegetation in the river landscape. Freshwater Biology 40: 497-516.

Tockner, K., F. Malard, and J. V. Ward. 2000. An extension of the flood pulse concept. Hydrological Processes 14: 2861-2883.

Tockner, K., D. Pennetzdorfer, N. Reiner, F. Schiemer, and J. V. Ward. 1999. Hydrological connectivity, and the exchange of organic matter and nutrients in a dynamic river-floodplain system (Danube, Austria). Freshwater Biology 41: 521-535.

Tockner, K., and F. Schiemer. 1997. Ecological aspects of the restoration strategy for a river-floodplain system on the Danube River in Austria. Global Ecology and Biogeography Letters 6: 321-329.

Tockner, K., F. Schiemer, and J. V. Ward. 1998. Conservation by restoration: The management concept for a river-floodplain system on the Danube River in Austria. Aquatic Conservation-Marine and Freshwater Ecosystems 8: 71-86.

Tockner, K., and J. A. Stanford. 2002. Riverine flood plains: present state and future trends. Environmental Conservation 29: 308-330.

Toth, L. A., S. L. Melvin, A. Arrington, and J. Chamberlain. 1998. Hydrologic manipulations of the channelized Kissimmee River ecosystem. Bioscience 48: 757-764.

Tremolieres, M., I. Eglin, U. Roeck, and R. Carbiener. 1993. The Exchange Process between River and Groundwater on the Central Alsace Floodplain (Eastern France) .1. The Case of the Canalised River Rhine. Hydrobiologia 254: 133-148.

Vandenbrink, F. W. B., J. P. H. M. Deleeuw, G. Vandervelde, and G. M. Verheggen. 1993. Impact of Hydrology on the Chemistry and Phytoplankton Development in Floodplain Lakes Along the Lower Rhine and Meuse. Biogeochemistry 19: 103-128.

Vandenbrink, F. W. B., and G. Vandervelde. 1994. Impact of Hydrology and Water Chemistry on Floodplain Lake Communities Along the Lower Rhine and Meuse. Water Science and Technology 29: 57-60.

Verhoeven, J. T. A., B. Arheimer, C. Q. Yin, and M. M. Hefting. 2006. Regional and global concerns over wetlands and water quality. Trends in Ecology & Evolution 21: 96-103.

Vericat, D., and R. J. Batalla. 2006. Sediment transport in a large impounded river: The lower Ebro, NE Iberian Peninsula. Geomorphology 79: 72-92.

Walling, D. E., Q. He, and A. P. Nicholas. 1996. Floodplains as Suspended Sediment Sinks. p. 399-440. In G. M. Anderson, D. E. Walling, & P. D. Bates (eds.), Floodplain Processes. Wiley, Chichester.

Ward, J. V. 1989. The 4-Dimensional Nature of Lotic Ecosystems. Journal of the North American Benthological Society 8: 2-8.

Ward, J. V. 1998. Riverine landscapes: Biodiversity patterns, disturbance regimes, and aquatic conservation. Biological Conservation 83: 269-278.

Ward, J. V., and J. A. Stanford. 1995. Ecological Connectivity in Alluvial River Ecosystems and Its Disruption by Flow Regulation. Regulated Rivers-Research & Management 11: 105-119.

Ward, J. V., K. Tockner, U. Uehlinger, and F. Malard. 2001. Understanding natural patterns and processes in river corridors as the basis for effective river restoration. Regulated Rivers-Research & Management 17: 311-324.

Weng, P., J. M. Sanchez-Perez, S. Sauvage, P. Vervier, and F. Giraud. 2003. Assessment of the quantitative and qualitative buffer function of an alluvial wetland: hydrological modelling of a large floodplain (Garonne River, France). Hydrological Processes 17: 2375-2392.

Whited, D. C., M. S. Lorang, M. J. Harner, F. R. Hauer, J. S. Kimball, and J. A. Stanford. 2007. Climate, hydrologic disturbance, and succession: Drivers of floodplain pattern. Ecology 88: 940-953.

STUDY I

Hydrologic and land-use change influence landscape diversity in the Ebro River (NE Spain)

Hydrologic and land-use change influence landscape diversity in the Ebro River (NE Spain)

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Abstract

The landscape dynamics (1927-2003) of one reach at the Middle Ebro River (NE Spain) was examined using aerial pictures and GIS techniques. Moreover, changes in the natural flow regime and anthropic activities within the river-floodplain system were investigated. Our results indicate that hydrological and landscape patterns have been dramatically changed during the last century as a consequence of human alteration of the fluvial dynamics within the studied reach, as well as the overall basin. The magnitude and variability of river discharge events have decreased, especially since 1981, and flood protection structures have disrupted the river floodplain connectivity. As a result, the succesional pathways of riparian ecotopes have been heavily modified because natural rejuvenation no longer takes place, resulting in decreased landscape diversity. It is apparent from these data that floodplain restoration must be incorporated as a significant factor into river management plans if a more natural functioning wants to be retrieved. The ecotope structure and dynamics of the 1927-1957 should be adopted as the guiding image, whereas hydrologic and landscape (dykes, raised surfaces) patters should be considered. Under the current socio-economic context, the more realistic option seems to create a dynamic river corridor reallocating dykes and lowering floodplain heights. The extent of this river corridor should adapt to the restored flow regime, although periodic economic investments could be an option if the desired self-sustained dynamism is not reached.

Keywords: Landscape analysis, riparian ecotope, flow regime, riparian succession, floodplain, Ebro.

Introduction

Linking landscape patterns and ecological processes is a common goal of landscape ecology (Forman and Godron, 1986). Landscape ecology holds the potential for developing a truly holistic perspective of river corridors by integrating structure, dynamics and function (Ward et al., 2002). The diversity of landscape units and their spatial distributions in pristine riverine landscapes are the result of geomorphological and biological processes and interactions operating across a wide range of spatio-temporal scales. As a consequence, it was through the interpretation of sequential landscape patterns that the primary drivers of the riverine landscape dynamics have been inferred in different studies (Miller et al., 1995; Hohensinner et al., 2004; Geerling et al., 2006; Whited et al., 2007). A full range of phenomena, ranging from catastrophic events to predictable mean flow, generate the fluvial dynamics and fluctuating hydrological connectivity that characterizes intact river-floodplain systems (Jungwirth et al., 2002). Riparian succession tends to drive aquatic environments toward terrestrial landscapes, but erosion and deposition during low-frequency floods truncate those successional pathways. As a result, in a diverse landscape which contains landscape units at every stage of succession, irregular and anticipated, events drive hydrogeomorphological functions and, in general, allow the system to remain stable (Amoros and Wade, 1996).

Anthropogenic alterations of floodplains often disrupt the intensity, frequency and timing of the natural disturbance regime that is key to the ecological integrity of riverine environments (Ward and Stanford, 1995). The need and/or desire for new space to develop, occupy and/or farm has greatly disturbed floodplains of small and large rivers alike. As a consequence, floodplains are among the most threatened ecosystems in the world despite their biological importance (Tockner and Stanford, 2002). At the Ebro River, in northeast Spain, the promotion of dam construction for irrigation purposes during the last century (Pinilla, 2006), resulted in the accelerated occupation of river margins and massive construction of flood protection structures. In the middle stretch of the Ebro, only about 4% of the floodplain is covered by natural vegetation (Ollero, 1992). Regato (1988) reported that natural vegetation had been strongly modified within the Ebro River study reach by alteration of the fluvial dynamics; this was later confirmed by Castro et al. (2001). Floodplain habitats, therefore, must be a critical component of river management for the Water Framework Directive to be successfully applied on the Ebro River.

To achieve the restoration of threatened river systems, a complete understanding of geomorphological and ecological processes is required (Kondolf, 1998). Such an understanding will serve as a basis to predict the potential effects of performing site-specific restoration either alone or in combination with flow allocation on a basin-wide scale. In this paper, the landscape dynamics of one study reach in the Middle Ebro River are investigated, as well as changes in the natural flow regime and anthropic activities, in order to achieve the next tasks: a) examine changes in hydrological and landscape patterns b) identify the factors that best explain the natural ecotope succession and c) propose a realistic restoration option with consideration of the landscape dynamics during the last century and the socio-economic context.

Methods

Study area

The study reach was located in the Middle Ebro River, NE Spain (Fig.1). This is the largest river in Spain (watershed area = 85,362 km2, river length = 910 km, average annual discharge to the Mediterranean Sea = 18138 Hm3) and is still geomorphologically active. The river meanders within this section (sinuosity = 1.39, bank slope = 0,050%), resulting in an average floodplain width of 5 km. Within the study reach, the mean discharge is 230 m3/s and the elevation ranges between 175 m a.s.l. in the river channel to 185 m a.s.l. at the base of the scarp. The estimated area that would be inundated by the 10-y flood event (3000 m3/s, 1927-2003) is 2230 ha, although only about 14% of that area would be inundated during a 1000 m3/s flood event (0,37 y return period, 1927-2003), and only 4% would be flooded by a river discharge of 500 m3/s. Upstream of the city of Zaragoza, the catchment area is 40,434 km2 and the dam-equivalent capacity is 1637.19 Hm3.

Hydrological analysis

Due to its main role in river-floodplain systems (Junk et al., 1989; Tockner et al., 2000), the pulsing of the river discharge, i.e. flood pulse, was used to characterize the hydrological patterns. It served as a basis to interpret landscape changes, although further analyses are required to interpret the direct effect of the components of the flow regime (magnitude, frequency, duration, timing, rate of change) over ecotope dynamics (see Poff et al., 1997). Daily average discharge, from 1927 to 2003 at Zaragoza, was provided by the Ebro River Basin Administration. This gauging station is located 12 km upstream of the study area and there are no major water diversions between the station and the study area; the discharge values, therefore, should

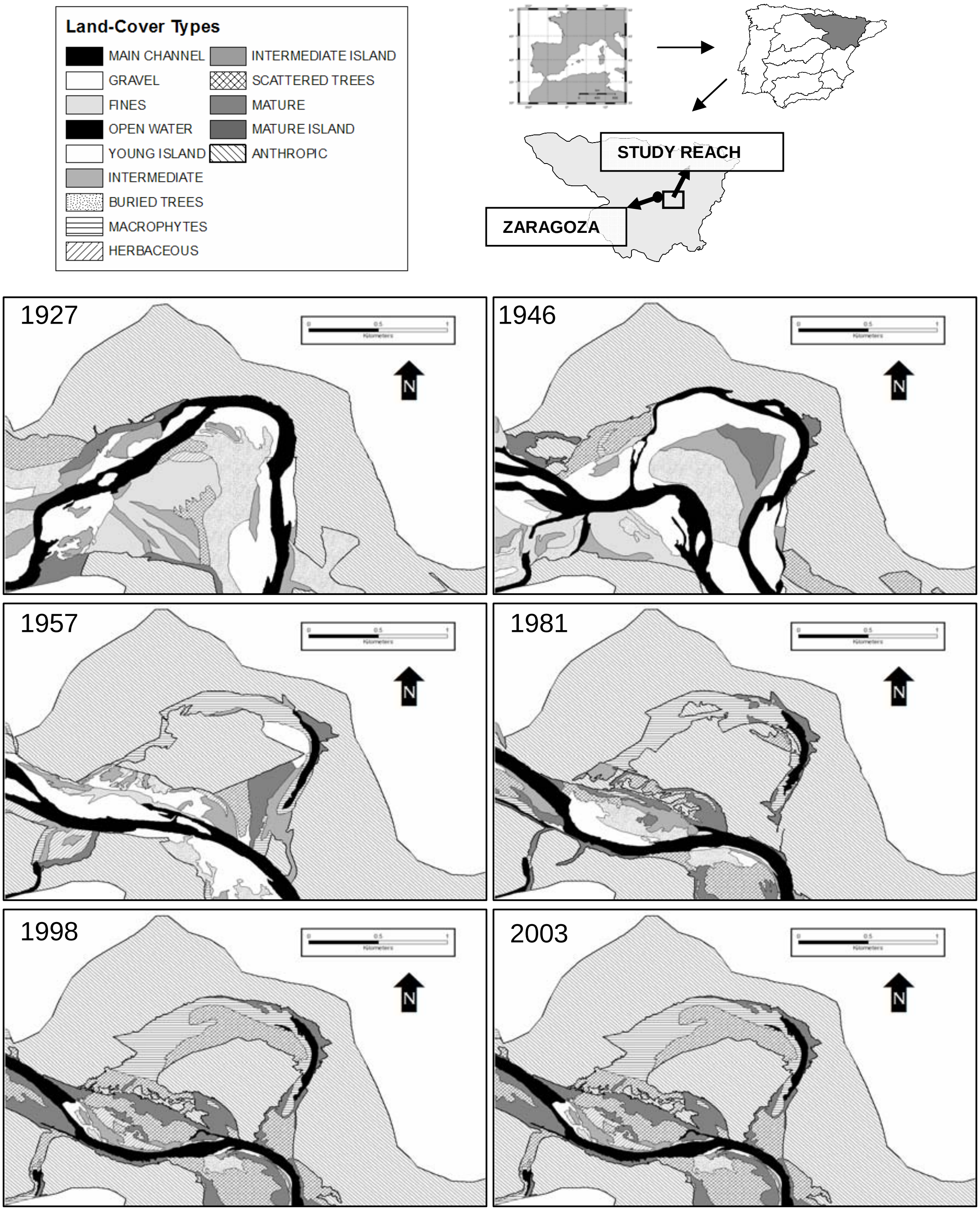

Figure. 1. Ecotope maps for a representative part of the study reach, including each of the six years considered in the study. River flow is from the upper left to lower right. There has been essentially no main channel migration between 1981 and 2003.. No significant changes were detected between 1998 and 2003.

be representative of conditions within the study area. A times series of flood events was generated from the original daily series. A flood event was defined as a series of one or more consecutive days with average daily discharge equal or higher than 600 m3/s. For each flood event the duration, peak discharge and cumulative discharge were recorded. A magnitude-frequency analysis of flood events was conducted using the partial duration series approach, with the purpose of determining the recurrence time of characteristic flood events. The partial duration series approach was preferred over the annual maximum series, which is the most widespread technique, due its superior mathematical properties and robustness (Beguería, 2005). The series of flood events were fitted to a Generalised Pareto distribution, which is the limit distribution for a series of events over a fixed threshold (Cunnane, 1973). In order to interpret ecotope dynamics through the hydrological patterns, data were separated and analyzed for three different periods (1927-1957; 1957-1981; 1981-2003), which coincide with the time-spans between aerial pictures. Bankfull discharge, an important parameter controlling channel and floodplain morphology, was defined as the flood event with an estimated recurrence time of 1.58 y (see Dury, 1981). Similarly, recurrence times for other river discharge values were also estimated for further inter-period comparisons. Finally, the mean annual discharge at the Zaragoza gauging station was also calculated.

Landscape analysis

Ecotope maps (Fig. 1) were generated from a set of aerial photographs (1927, 1946, 1957, 1981, 1998 and 2003) to perform a landscape transition analysis using GIS techniques. The 1946, 1957 and 1981 photographs were black-and-white at different resolution scales (1:40000, 1:33000 and 1:18000, respectively). They were rectified and georeferenced using LPS® 9.1 (ERDAS Imagine 9.1®). The 1927 images were supplied by the Ebro River Basin Administration as rectified aerial photographs (1:10000) and georeferencing was performed with ArcGis 9.2®. Both maps and aerial pictures had been previously scanned at 600 dpi, yielding raster images with a pixel resolution from 1 to 2 m. Positional accuracy (n = 20) in the studied floodplain averaged 5 m for all georeferenced images. Finally, 1998 and 2003 aerial pictures were supplied by the Aragon Regional Government as georeferenced images with a 1.0 and 0.5 m pixel resolution, respectively.

Three years of field campaigns served as a basis for the identification of ecotope types (Table 1). Landscape units were then delimited and classified following a simple interpretation-key, which was created using texture, colour, tree density, vertical structure, position in the landscape or previous channel migration dynamics. Landscape data were digitized using ArcGis 9.2® with a fixed scale of 1:3000. When possible, a stereoscope was used to exploit the original quality and vertical information of the aerial photos. All patches smaller than 64 m2 were eliminated and vector maps were rasterised to a 10 x 10 m grid using ArcGis 9.2®.

To explore the relationship between landscape structure and human modification of river-floodplain interactions, ecotope maps were progressively truncated by increasing the distance to the main channel by 100 m, up to 1000 m, and every 500 m from 1000 to 2500 m. This final buffer distance included the entire 10 y floodplain, which has been considered the reference area for the landscape metrics. Delineation of this reference area was refined by the Ebro River Basin Administration using remote sensing data and ground-truthing during the February 2003 flood, which peaked at 2 988 m3/s at the Zaragoza gauging station (Losada et al., 2004). For all buffers considered in this study, Fragstats 3.3 (McGarigal and Marks, 1995) was used to calculate the area and percentage of land occupied by each ecotope category (CA and PLAND), as well as ecotope diversity using the Shannon Index (SHDI).

To examine ecotope change, transition matrices and maps were produced for each time span using IDRISI Kilimanjaro® (CrossTab). A general ecotope succession model (Fig. 2) was then created from the interpretation of transition matrices and previous research on vegetation dynamics

Ecotope	Successional Stage	Description
Main Channel	**-**	**Area occupied by main channel**
Gravel	Initial	Covered by gravel, adjacent to the main channel
Fines	Initial	Covered by fine substrate, adjacent to the main channel
Open water	Initial	Flooded areas with no emergent vegetation
Young Island	Initial	Located in-shore, covered by gravels
Intermediate	Intermediate	Closed canopy (>75%), young individuals
Buried trees	Intermediate	Coarse substrate, clustered young trees
Macrophytes	Intermediate	Covered by emergent vegetation
Herbaceous	Intermediate	Absence of trees, not adjacent to the main channel
Inter. Island	Intermediate	Located in shore, not covered by gravels or mature trees
Scattered Trees	Mature	Fine substrate, clustered mature trees.
Mature	Mature	Closed tree canopy (>75%), mature individuals
Mature Island	Mature	Located in shore, covered by mature trees
Anthropic	-	Agricultural fields or poplar groves

Table 1. Ecotope types used to define landscape units.

(Braun Blanquet and de Bolós, 1987; Regato, 1988). Every ecotope transition was classified either as a Natural transition (succession (SUC), rejuvenation (REJ) or stability (STA) according to the succession model (Fig.2)) or as human-affected transition (towards the “Anthropic” type). Using the cartographic ecotope data from the previous year, we determined how ecotope types developed from the initial patchwork. For this analysis, the importance (%) of SUC, REJ and STA during the analysed snap-shots was represented in triangular ternary plots discarding the human-affected transitions , as recently constructed by Geerling et al. (2006). Conversely, using the following year as the reference point we identified which fraction of the final patchwork belonged to each successional process.

Results

Hydrological analysis

Our analysis revealed a clear decrease in the mean annual discharge at the Zaragoza gauging station since 1981. Although discharge showed an apparent increase in the first two periods (1927-57 and 1957-1981), rising from 7930 to 8720 Hm3/y, it fell to 5834 Hm3/y during the last twenty years. The magnitude and frequency of floods in the Ebro River also decreased since 1981 (Fig. 3). Similar-magnitude discharges were estimated to occur with similar frequencies from 1927 to 1981, after which their periodicity declined. For example, a 3000 m3/s event, which served to delimit the floodplain area, had a recurrence time of 8 y and 10 y in the periods 1927-1957 and 1957-1981, respectively. From 1981 and 2003, however, the frequency of an event of that magnitude decreased, with a recurrence time of 60 years. Similarly, the bankfull discharge dropped slightly from 1980 to 1917 m3/s between the first and the second time periods, but diminished substantially to 1410 m3/s in the 1981 to 2003 period. The number of flood events in which the peak exceeded the bankfull discharge has decreased in since 1981 (Table 2) although no real drop was visible in the per year occurrence. The magnitude and duration of these flood events were higher before 1981, although its relative proportion with respect to the total number of floods has increased over time. The number and frequency of floods that peaked below bankfull discharge have progressively dropped over the last 80 years, going from 206 sub-peak events in the 1927-1957 period (7.10 events per year), to only 90 such events in the 1981-2003 period (4.09 events per year) (Table 2). However,

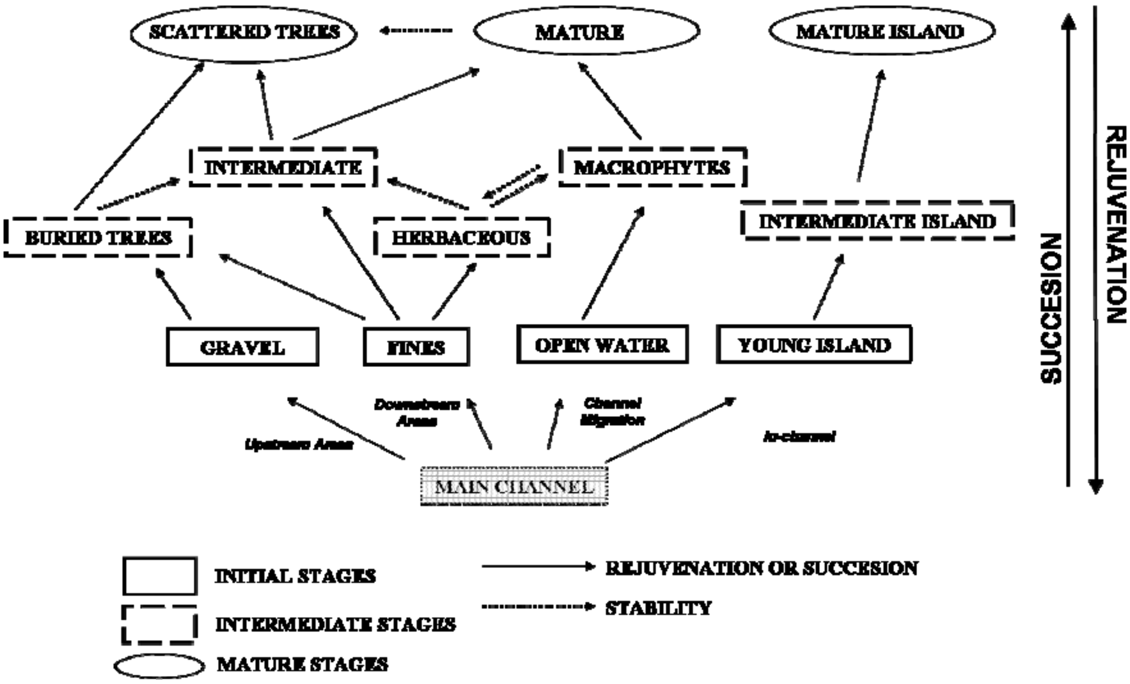

Figure. 2. Ecotope succession scheme.

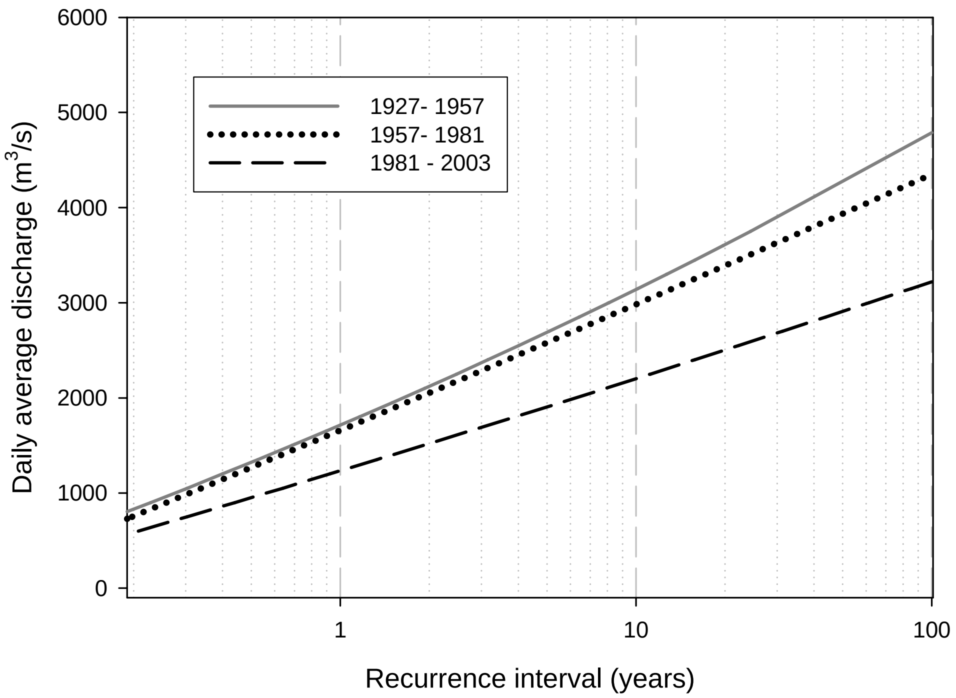

Figure. 3. Magnitude-frequency plots illustrating reduced frequency of high-discharge events in more recent years.

Period	Peak Discharge (m3/s)	Events		Duration (d)		Cumulative Discharge (m3/s)	
		total	**per year**	**total**	**per event**	**total**	**per event**
1927-1957	600 < x > 1980	206	7.10	811	3.94	738098.95	3583.00
	x > 1980	21	0.72	335	15.95	437880.21	20851.44
	TOTAL	227	7.82	1146		1175979.16	
1957-1981	600 < x > 1957	146	5.84	669	4.58	593994.61	4068.46
	x > 1957	20	0.80	310	15.50	406952.18	20347.61
	TOTAL	166	6.64	979		1000946.79	
1981-2003	600 < x > 1410	90	4.09	303	3.37	246465.12	2738.50
	x > 1410	16	0.73	161	10.06	180496.64	11281.04
	TOTAL	106	4.82	464		426961.76	

Table 2. Number, duration and accumulated mean discharge of flood events at the Zaragoza gauging station. Data were analyzed separately for the three temporal periods. Flood events were categorized according to the estimated bankfull discharge.

duration and magnitude of sub-peak events have oscillated, reaching a maximum (4.58 d per event) in the 1957-1981 period, and a minimum (3.37 d per event) in the most recent period, showing no clear trends.

Landscape analysis

Ecotope maps show how drastically the landscape structure has changed from 1927 to 2003 (Fig. 1). Ecotope diversity has decreased over that same period. The Shannon Diversity index (H) of the entire floodplain area, which corresponds to the 2500 m buffer, dropped from 1.78 in 1927 and 1946 to 1.08 in 1998 and 2003 (Fig. 4). In 1981, this index was slightly lower than in 1998. In addition, there has been a spatial change in ecotope diversity within the floodplain relative to proximity of the main river channel. Prior to 1981, ecotope diversity peaked at a distance of 300 m from the river bank and then decreased with increasing distance from the river; the 1946 spectra shows a secondary, though slight, peak at 900 m. However, the 1981, 1998 and 2003 data show maximum ecotope diversity at just 100 meters from the main channel, followed by a rapid and steep drop.

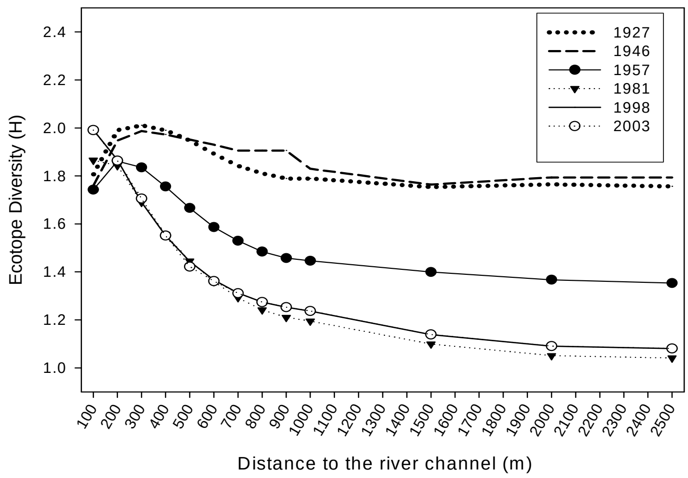

Figure. 4. Ecotope diversity as a function of distance to the river channel. Note that 1998 and 2003 plots are superimposed

ECOTOPE	AREA (%)											
	Floodplain Area						Area occupied by Natural Ecotopes					
	1927	1946	1957	1981	1998	2003	1927	1946	1957	1981	1998	2003
Main Channel	8.24	9.08	6.83	6.63	5.25	5.33	16.29	17.36	19.93	27.98	19.84	20.16
Gravel	7.96	10.05	6.84	2.60	1.30	1.18	16.08	19.19	19.97	10.97	4.90	4.45
Fines	2.63	3.13	0.40	0.38	0.60	0.60	5.31	5.98	1.15	1.59	2.26	2.26
Open Water	0.48	1.62	0.81	1.03	0.69	0.73	0.98	3.09	2.35	4.33	2.64	2.75
Young Island	0.64	1.66	0.16	0.46	0.17	0.17	1.65	3.17	0.47	1.93	0.66	0.66
INITIAL	19.95	25.54	15.04	11.1	8.01	8.01	40.31	48.79	43.87	46.8	30.3	30.28
Intermediate	5.41	5.07	4.59	2.43	1.22	1.28	10.95	9.68	13.42	10.23	4.62	4.84
Buried Trees	6.18	3.02	2.68	1.22	0.30	0.35	12.50	5.78	7.79	5.13	1.14	1.32
Macrophytes0	7.01	4.13	4.58	1.82	2.24	2.25	14.17	7.87	13.36	7.69	8.46	8.50
Herbaceous	0.08	0.08	0.73	0.19	0.52	0.51	0.14	0.16	2.14	0.79	1.97	1.93
Inter. Island	0.38	0.00	0.01	0.05	0.03	0.03	0.77	0.00	0.03	0.21	0.11	0.11
INTERMEDIATE	19.06	12.3	12.59	5.71	4.31	4.42	38.53	23.49	36.74	24.05	16.3	16.7
Scattered Trees	4.38	10.79	4.30	2.90	7.91	7.89	8.83	20.62	12.53	12.24	29.91	29.81
Mature	6.09	3.71	2.35	3.95	6.02	5.94	12.32	7.09	6.86	16.65	22.80	22.51
Mature Island	0.00	0.00	0.00	0.06	0.18	0.18	0.00	0.00	0.00	0.24	0.69	0.69
MATURE	10.47	14.5	6.65	6.91	14.11	14.01	21.15	27.71	19.39	29.13	53.4	53.01
Anthropic	50.53	47.66	65.73	76.29	73.56	73.56						
TOTAL	100	100	100	100	100	100	100	100	100	100	100	100

Table 3. Area (%) covered by each ecotope in each year. Percentages are calculated for the total floodplain area (2230 Ha) and for the area occupied by natural ecotopes. Sub-totals by succesional stage (see Table 1) are also displayed.

Ecotope	1927-1946	1946-1957	1957-1981	1981-1998	1998-2003
Main Channel	0.67	1.22	1.89	0.62	0.00
Gravel	0.13	27.56	19.97	4.37	0.00
Fines	0.18	70.05	51.18	2.60	0.00
Open Water	0.31	5.98	6.71	1.88	0.00
Young Island	0.00	1.99	0.00	0.00	0.00
Intermediate	3.86	38.15	52.97	4.71	0.00
Buried Trees	14.65	52.03	35.33	2.97	0.00
Macrophytes	8.98	69.16	69.38	1.96	0.00
Herbaceous	0.00	60.98	82.41	0.24	0.00
Inter. Island	0.00	-	0.00	0.00	0.00
Scattered Trees	16.64	64.39	88.13	4.08	0.00
Mature	17.15	43.37	81.76	7.62	0.00
Mature Island	-	-	-	0.00	0.00
TOTAL (Ha)	73.06	449.80	326.07	17.619	0

Table 4. Percentage of each natural ecotope category converted to “Anthropic” in each transition category. Also the entire area of natural ecotopes converted to “Anthropic” in each transition is shown.

Elongated meanders were present in the 1927 maps, but these oxbow channels were cut-off before 1946 (Fig. 1). Lateral accretion caused the main channel to migrate between 1946 and 1981, and established its current location. The area of the main channel decreased over the study period (8.24% in 1927 to 5.33% in 2003), whereas its relative importance within the natural ecotopes area increased (16.29% in 1927 to 20.16% in 2003) (Table 3). In contrast to the aerial decrease in most of the natural ecotopes, human-occupation of the river space has markedly increased in importance, especially between 1946 and 1981 (Table 4). This change appears true not only for the outer margins of the floodplain, where incremental impacts would be expected, but also the natural riparian corridor, adjacent to the main channel, which has narrowed considerably (Fig. 1).

The ecotope transitions were grouped into the three time periods in accordance with the hydrological analysis (Fig. 5): a) 1927-1957, transitions 1 and 2, b) 1957-1981, transition 3 and c) 1981-2003, transitions 4 and 5. Natural ecotopes at different successional stages (Table 1) evolved in a distinctive manner which influenced their area coverage in the final year of each transition (Table 3). During the first phase, channel avulsion after cut off of the meanders (Fig. 1, 1927-1946) increased the area of the ecotopes during the initial stages (Table 3). However, younger ecotopes decreased in importance between 1946 and 1957, when conversion of natural landscapes to agricultural fields (Table 4) resulted in marked changes in the “gravel” and “fines”. Ecotope rejuvenation (REJ) and stability (STA), on one hand, were in balance with succession (SUC), which was even impeded for the “gravel” and “fines” patches in the 1927-1946 transition (Fig. 5). For the intermediate stages, distinct successional pathways (Fig. 2) resulted in a different dynamics. “Intermediate” and “buried trees”, normally located adjacent to the main channel, showed high REJ percentages while their areal coverage decreased between 1927 and 1946. This trend prevailed until 1956, although their areas remained stable despite anthropic pressure (Table 4). However, succession was balanced with stability during this same period, as evidenced by an approximate two-fold decrease in “macrophytes” area. The remaining patches were highly affected by conversion to agricultural fields along during 1946-1957, mainly due to loss in the outer floodplain (Table 4, Fig. 1). The area occupied by emergent vegetation remained stable between 1946 and 1957 . For “herbaceous” and “intermediate island”, no clear trends were detected. Finally, even though patches of the various mature stages were renewed progressively between 1927 and 1957, their areal coverage did not decrease proportionally (e.g., “mature”), and, in some cases, even increased (e.g., “scattered trees”).

From 1957 to 1981, trends were similar to those in the first time period for “open water” and “gravel”, but SUC became dominant for “fines” and “young island”. During this period, only the “gravel” area diminished (Table 4). For intermediate stages, SUC also emerged as the dominant process, although REJ was also important. Their areas decreased due to the high rates restricted the establishment of “scattered of conversion to “anthropic” and the low stability (Fig. 5). Referring to mature ecotopes, rejuvenation and anthropization trees” and “mature”. However, those ecotopes became dominant, with the latter (“mature”) even increasing its importance. During the last period (1981-2003), the areas containing ecotopes at initial and intermediate stages decreased, with the exception of macrophytes. In contrast, “mature” and “scattered trees” accounted for about 50% of the floodplain (Table 3). Young islands existing in 1981 were destroyed and re-established in another location along the main channel (Fig. 1), while “mature islands” first appeared in 1998. All ecotope types showed a slight trend towards STA between 1981 and 1998, with the exception of “island” ecotopes. During that transition, REJ became nearly nonexistent (Fig. 5). Finally, all ecotopes showed increasing signs of stability after 1998, despite the potential erosive effect of a 60 y flood (1981-2003) in February 2003.

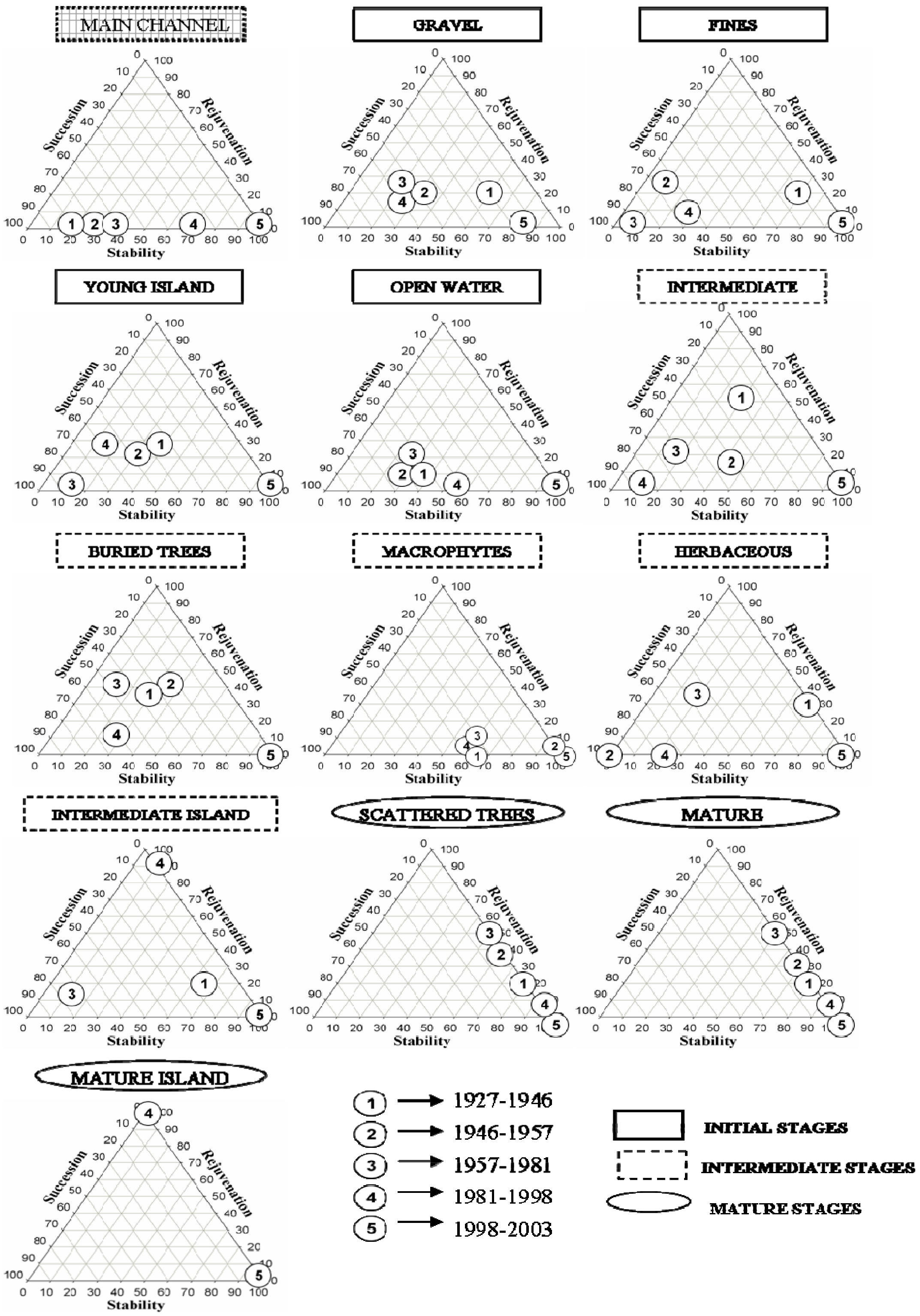

Figure. 5. Ternary plots for ecotope succession, rejuvenation or stability, considering the fraction of the ecotope not converted to "Anthropic". "Intermediate Island" did not exist in 1957 while "Mature Island" appeared in 1981.

Ecotope	1927-1946	1946-1957	1957-1981	1981-1998	1998-2003
Main Channel	0.00	0.00	0.00	0.00	0.00
Gravel	31.54	41.05	45.66	36.64	0.00
Fines	5.23	49.94	44.21	35.54	0.00
Open Water	29.31	47.52	41.93	13.20	0.00
Young Island	18.35	39.93	29.77	85.03	0.00
Intermediate	42.73	68.53	64.87	59.08	5.42
Buried Trees	45.88	87.21	87.20	37.68	13.32
Macrophytes	0.66	34.24	16.83	14.92	0.00
Herbaceous	2.65	79.25	90.13	71.51	0.00
Inter. Island		100.00	18.89	82.10	0.00
Scattered trees	38.74	28.07	77.87	31.96	0.00
Mature	49.77	49.62	85.73	38.65	0.00
Mature Island	-	-	100.00	100.00	0.00

Table 5. Percentage of each ecotope remaining from an earlier natural ecotope at the end of each transition

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Discussion

Changes in hydrological and landscape patterns

Over the last century, the natural flow regime in the Middle Ebro River has been modified by progressive river flow regulation and anthropization of the catchment area. However, it has been since 1981 that discharge magnitude and variability have markedly decreased (Fig. 3, Table 2). The mean annual discharge within the study reach has declined approximately 30% since 1981, coinciding with the decrease of bankfull discharge. Various researchers have suggested that this phenomenon has been caused by the progressive increase of evapotranspiration due to higher temperatures, reforestation of abandoned agricultural fields in mountainous areas, increase in reservoir water storage (volume and surface area) and the expansion of irrigated farmlands (Ibanez et al., 1996; Ollero, 2007). In basin areas upstream of the study reach, a sharp increase in the total equivalent capacity of reservoirs occurred between 1950 and 1980, in parallel with the expansion of irrigated land. However, the emphasis on agricultural production since the 1980's was driven to a large degree by the cultivation of water-hungry crops such as rice, fruits or vegetables (Frutos et al., 2004). In addition to this increased demand, precipitation peaked during the 70´s in nearly all of the Ebro Basin (Abaurrea et al., 2002), which may have helped to dampen the effect of human impacts on the system. Flood events, flow and flood pulses (see Tockner et al., 2000) have shown distinct patterns of change in their frequency, magnitude and duration (Table 2). Since 1981, events above the bankfull discharge decreased in magnitude and duration while those below bankfull discharge decreased only in number. The management of the largest reservoirs for irrigation purposes might explain those trends. In winter, the number and magnitude of floods is dampened using the existing flood control infrastructure in order to store water for summer, when irrigation demand is high. Given the winter storage goals, the capacity of flood-control systems to mitigate large floods in early spring, when the snowmelt occurs, is minimal. This has been previously described for dams located in the Pyrenees, which are often at or near capacity at the onset of spring floods (Lopez Moreno et al., 2002).

Landscape structure seemed to adapt to changes in the flow regime, although such adjustment was altered by human disruptions of river-floodplain interactions. Both human occupation of the river space and dyke construction have accelerated the evolution towards a less diverse landscape, and modified the spatial patterns of landscape diversity (Fig. 4). The highest ecotope diversity (ED) was observed prior to the 1960´s because it was during that time that construction of major flow regulation infrastructure began that would truncate the successional pathways. Moreover, flood events maintained a diverse array of landforms in areas adjacent to the main channel, what is not reflected in the spectra after 1957 might due to the effect of dykes. Diversity peaked at 300 m from the river channel (Fig. 4), where the progressive dominance of “anthropic”, intermediate and mature ecotopes (Fig. 2) forced a decline in diversity. Prior to 1957, large natural patches in the outer floodplain (Fig. 1) caused diversity to be similar in 1927 and, 20 years later, in 1946. However, agricultural expansion in the floodplain substantially lowered diversity only 10 years later in 1956. Conversion from natural to anthropic ecotopes proceed almost entirely from 1946 to 1981 (Table4), resulting in a two fold increase of the area covered by human-managed ecotopes (Table3). After abandonment, some of those human-manages patches have been covered by natural vegetation, what explains the slight increase of ecotope diversity since 1981 (Fig.4).

The drop in ecotope diversity after the 1960’s might be explained, in part, by changes in hydrology, which promoted the decrease of ecotopes at the initial successional stages (Table 3). However, the magnitude of river flow and fluctuations in river discharge between 1957 and 1981 did not greatly differ from the previous period of 1927 to 1957 (Fig. 3). Therefore, it was the extensive implementation of flood protection structures along the river banks that was the most likely factor disrupting river-floodplain interactions. Although defences have been constructed along the Ebro River for centuries (Ollero, 2007), within the study reach almost all modern and effective flood protection structures were built between 1960 and 1980 (Ollero, 1992), and they are reinforced after every large flood. Thus, the strong dominance of mature ecotopes since the 1990´s has been in response to the synergic effect of flow regulation and flood protection which has severely reduced natural changes within the floodplain. In addition, riverbed incision from dam construction probably counteracts the interaction between the main channel and its adjacent ecotopes during floods. Vericat and Batalla (2006) reported a river bed incision of 3 cm per year for the lower Ebro.

Natural ecotope succession

Human alteration of river-floodplain interactions, which occurred sequentially, prevailed over the natural drivers of floodplain dynamics since 1957. This promoted a different ecotope dynamics, as well as distinct initial conditions, for each time period considered in this study: a) Channel Migration (1927-1957): unmodified flow regime and absence of flood protection, high conversion to anthropic landcovers took place at least since 1946, b) Vertical Accretion (1957-1981): unmodified flow regime but establishment of flood protection (Ollero, 1992), high rates of conversion to anthropic landcovers and c) Homogenization (1981-2003): modified flood regime and dyke construction, conversion to anthropic ecotopes with a great reduction in natural ecotopes.

During the first phase (1927-1957), the river-floodplain interactions during flow pulses not only stabilized part of the area occupied by initial and intermediate ecotopes, but also compromised the ongoing succession through main channel migration. Flood scouring forced larger patches of mature ecotopes to be located at the outer floodplain (Fig. 1), whereas flood events were sufficiently robust to rejuvenate mature patches located adjacent to the main channel

(Fig. 5). Between 1946 and 1957, the elevated river-floodplain connectivity allowed for rapid readjustments to disturbances within the watershed, which explains the differences between initial and intermediate ecotope areas between 1946 and 1957 (Table 3) and the percentage belonging to earlier stages (Table 5).

During the second phase (1957-1981), river-floodplain interactions were strong enough to allow channel migration before the main channel adjusted to its "straitjacket" (sensu Lamers et al., 2006), and therefore, rejuvenation occurred at every successional stage (Fig. 5). The intense conversion to human-managed ecotopes restricted natural patches to the river corridor (Fig. 1), while lateral accretion was progressively constrained by dyke construction. As a consequence, succession was probably accelerated by higher vertical accretion rates. Steiger et al. (2001) described this trend for a riparian area over a 30-y period in the Garonne river.

During the last period (1981-2003), the synergic effect of a modified flow regime and flood protection impeded ecotope rejuvenation, with the exception of in-channel ecotopes (Fig. 5). Between 1981 and 1998, the effect of non-erosive floods caused succession to proceed for the initial and intermediate ecotopes, increasing the importance of mature ecotopes at the end of the period (Table 3). The last examined transition period (1998-2003) revealed strong system stability (Fig. 5), despite the potential effects of a 3000 m^3/s (60 y; 1981-2003) in February 2003. According to Amoros and Wade (1996), tremendous quantities of external energy are required to revert succession because in mature ecotopes, which accounted for 50% of the natural ecotopes in 1998 (Table 3), succession is driven by autogenic processes. Indeed, only bank erosion at localized points was detected between 1998 and 2003.

Restoration options for the Middle Ebro floodplains

As described by Stanford et al. (2005), floodplain elements tend to persist in natural river systems, although their spatial distribution shifts over time due to flow-related changes. At the study reach this was observed until 1957, as shown by the proportion of ecotopes at different successional stages (Table 3, right). Under such natural conditions, erosion during low-frequency floods truncate successional pathways (Amoros and Wade, 1996). Peaks at the diversity spectra (Fig.4) indicate that it occurred at areas closed to the main channel until 1957, before dykes started to be set. It seems, therefore, that hydrological and landscape patterns prior to 1957 allowed natural ecotope dynamics. Since 1957, those patterns have been markedly modified, and so river-floodplain interactions responsible to maintain natural ecotope dynamics. Consequently, the ecotope diversity and dynamics observed between 1927 and 1957 is considered by the authors as a valid reference situation if a more natural functioning of the river-floodplain system wants to be achieved through ecological restoration.

To plan and accomplish this, landscape and hydrological constraints must be considered. Existing landscape disturbances (dykes, vegetation encroachment, and raised surfaces) will limit the restoration success if the natural flow regime is recovered to a more historical condition. Although surface and groundwater connectivity would be enhanced, the erosive effect of floods would be strongly counteracted by artificial dykes constructed at convex banks and vegetation encroachment at concave banks. Also, accretion would be accelerated in the outer floodplain, as it has been observed during the period 1957-1981. Ecotope dynamics could increase in the main channel and adjacent areas. Given these likely conditions, the overall ecotope diversity would not substantially change, although it could increase along the riverbanks. Similarly, hydrological patterns will limit the restoration success if floodplain topography is modified to enhance connectivity. A dynamic corridor could be created if dykes are removed or re-located and floodplain height is lowered, but the current flow regime is not adequate to ensure self-sustained processes over the entire floodplain area. Moreover, the main channel will probably adjust to new conditions after a certain period of time, as it was observed during 1946-1957.

To deal with such landscape and hydrologic constraints, we strongly recommend that river management decisions and scientific knowledge should be integrated across scales, as pointed out by Hughes et al. (2001). At basin scale, alternative strategies would need a more integrated use of natural resources, consisting primarily of soil and water (Comin, 1999). Beyond that, however, the current land uses within the Ebro Basin territory must also be integrated into a management plan, as performed for other threatened floodplains (Rhode et al., 2006; Hale and Adams, 2007). At present, 80 % of the Ebro water demand is diverted for agriculture and farming (Frutos, 2004), accounting for about 40% of the mean annual discharge. This trend is not likely to be changed because of further irrigation development is planned (www.chebro.es). At reach scale, public reclamation of agricultural lands for restoration purposes seems possible due to the current socio-economic context. The average age of landholders has increased through time, and a high percentage of the crops are only profitable due to agricultural subsidies. With regards to flood protection structures, decisions concerning on dyke reallocation rely entirely on the Ebro Basin administration.

Under this scenario, the more realistic option is to create a dynamic river corridor where the river is the engine behind system maintenance. Within this corridor, landscape patterns should mimic those observed in 1927 and 1957 if restoration succeeds. With regard to flow regime, it would be unrealistic to attempt to restore the magnitude of river discharge, also because the forecast of climate and global changes establishes a potential reduction of the river discharge in the Ebro basin (Lopez-Moreno et al., 2008). Instead, efforts should focus on mimicking the other components of the flow regime (frequency, duration, timing, rate of change) since that will more likely contribute to sustain the riparian dynamics (Poff et al., 1997; Bendix and Hupp, 2000; Hughes and Rood, 2003; Stromberg et al., 2007). To reach the desirable self-sustainable ecotope dynamism, the extent of the river corridor should adapt to the restored flow regime. As exemplified by Greco et al. (2007), geomorphological dynamics are necessary to maintain such dynamics. The current landscape constraints (dykes, raised surfaces) have to be eliminated within the dynamic river corridor. It should be achieved by an initial economic investment, which will be required to be periodic if the self-sustained dynamism is not reached. This strategy has been proposed as a valid compromise between the need for flood protection and the growing demand for ecosystem rehabilitation in highly regulated rivers (Baptist et al., 2004). Additional monitoring, on a decadal–scale, will be required to evaluate the success of previously defined restoration plans, redefining them if key parameters change (Hughes et al., 2005).

Conclusions

Flow regulation, human occupation and construction of flood protection structures have modified landscape structure and dynamics in the middle Ebro River. At present, the fluvial landscape is less diverse and dominated by mature stages and anthopic ecotopes, river-floodplain interactions are counteracted by dykes, and hydrological patterns are different, in terms of pulses of the river discharge, to those observed prior to river regulation (1957-1957). It seems, therefore, that a more natural functioning of the river-floodplain system should be achieved through ecological restoration. To accomplish this goal, ecotope diversity and dynamics observed between 1927 and 1957 is a valid reference situation. When implementing restoration, managers should consider the current hydrological and landscape constraints (dykes, vegetation encroachment, and raised surfaces) and the socio-economic context at basin and reach scale. The more realistic option is creating a dynamic river corridor whose extent should adapt to the restored flow regime. An initial economic inversion is necessary to reallocate dykes and lowering floodplain height; however, it might be required periodically if self-sustained restoration is not achieved within this dynamic river corridor.

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References

Abaurrea, J., Asín, J. , Centellea, A., 2002. Spatio-temporal characterization of annual precipitation in the Ebro Basin, in: J.A. Guijarro, M. Grimalt, M. Laita , S. Alonso (Eds.), El agua y el Clima. Publicaciones de la Asociación Española de Climatología, 113-124. (in Spanish).

Amoros, C. , Wade, P.M., 1996. Ecological succesions, in: G.E. Petts , C. Amoros (Eds.), Fluvial Hydrosystems. Chapman & Hall, London, pp. 211-241.

Baptist, M.J., Penning, W.E., Duel, H., Smits, A.J.M., Geerling, G.W., van der Lee, G.E.M. et al.., 2004. Assessment of the effects of cyclic floodplain rejuvenation on flood levels and biodiversity along the Rhine river. River Res. Appl, 20, 285-297.

Begueria, S., 2005. Uncertainties in partial duration series modelling of extremes related to the choice of the threshold value. J. Hydrol. 303, 215-230.

Bendix, J. , Hupp, C.R., 2000. Hydrological and geomorphological impacts on riparian plant communities. Hydrol. Process. 14, 2977-2990.

Braun-Blanquet, J. , de Bolòs, O., 1987. The vegetation communities of the Ebro valley and its dynamism, Delegación Medio Ambiente - Ayuntamiento de Zaragoza, Zaragoza. (in Spanish).

Castro, P., Guerrero, J. , Muñoz, M.A., 2001. Restoration planning of the riparian forest at the Galachos Natural Reserve (Zaragoza).Consejo de Proteccion de la Naturaleza de Aragon, Zaragoza. (in Spanish).

Comin, F.A., 1999. Management of the Ebro River Basin: Past, present and future. Water Sci. Technol. 40, 161-168.

Cunnane, C., 1973. A particular comparison of annual maxima and partial duration series methods of flood frequency prediction. J. Hydrol. 18, 257-271.

Dury, G.H., 1981. Magnitude-Frequency analysis and Channel Morphology, in: M. Morisawa (Eds.), Fluvial Geomorphology. Allen and Unwin, London, pp. 91-121.

Forman, R.T.T. , Godron, M., 1986. Landscape Ecology, Jhon Wiley and Sons, New York.

Frutos, L.M., Ollero, A. , Sánchez Fabre, M., 2004 Characterization of the Ebro river and its catchment and its hydrological fluctuations, in: A. Gil Oncina (Eds.), Alteración de los regimenes fluviales peninsulares. Fundación Caja Murcia, Alicante, pp. 223-280. (in Spanish).

Geerling, G.W., Ragas, A.M.J., Leuven, R., van den Berg, J.H., Breedveld, M., Liefhebber, D. et al.., 2006. Succession and rejuvenation in floodplains along the river Allier (France). Hydrobiol. 565, 71-86.

Greco, S.E., Fremier, A.K., Larsen, E.W. , Plant, R.E., 2007. A tool for tracking floodplain age land surface patterns on a large meandering river with applications for ecological planning and restoration design. Landscape Urban Plann 81, 354-373.

Hale, B.W., Adams, M.S., 2007. Ecosystem management and the conservation of river-floodplain systems. Landscape Urban Plann 80, 23-33.

Hohensinner, S., Habersack, H., Jungwirth, M. , Zauner, G., 2004. Reconstruction of the characteristics of a natural alluvial river-floodplain system and hydromorphological changes following human modifications: The Danube River (1812-1991). River Res. Appl, 20, 25-41.

Hughes, F.M.R., Adams, W.M., Muller, E., Nilsson, C., Richards, K.S., Barsoum, N. et al., 2001. The importance of different scale processes for the restoration of floodplain woodlands. Regul. Rivers Res. Manag. 17, 325-345.

Hughes, F.M.R., Colston, A. , Mountford, J.O., 2005. Restoring riparian ecosystems: The challenge of accommodating variability and designing restoration trajectories. Ecol. Societ. 10, 12.

Hughes, F.M.R. , Rood, S.B., 2003. Allocation of river flows for restoration of floodplain forest ecosystems: A review of approaches and their applicability in Europe. Environ. Manag. 32, 12-33.

Ibanez, C., Prat, N. , Canicio, A., 1996. Changes in the hydrology and sediment transport produced by large dams on the lower Ebro river and its estuary. Regul. Rivers Res. Manag. 12, 51-62.

Jungwirth, M., Muhar, S. , Schmutz, S., 2002. Re-establishing and assessing ecological integrity in riverine landscapes. Freshw. Biol. 47, 867-887.

Junk, W.J., Bayley, B. , Sparks, R.E., 1989. The flood pulse concept in river-floodplain systems. in D.P.Dodge (ed) Proc. Intern. Large River Symposium. Special Issue of J.Canad. Fish and Aquat. Sci. 106, 11-27.

Kondolf, G.M., 1998. Lessons learned from river restoration projects in California. Aquat. Conserv. Marine Freshw. Ecosyst. 8, 39-52.

Lamers, L.P.M., Loeb, R., Antheunisse, A.M., Miletto, M., Lucassen, E.C.H.E.T., Boxman, A.W. et al.., 2006. Biogeochemical constraints on the ecological rehabilitation of wetland vegetation in river floodplains. Hydrobiol. 565, 165-186.

Lopez-Moreno, J.I., Begueria, S. , Garcia-Ruiz, J.M., 2002. Influence of the Yesa reservoir on floods of the Aragon River, central Spanish Pyrenees. Hydrol. Earth. Syst. Sci. 6, 753-762.

Lopez-Moreno, J.I., Beniston, M., García-Ruiz, J.M., 2008. Environmental change and water management in the Pyrenees: Facts and future perspectives for Mediterranean mountains. Glob. and Planet. Change. 61, 300-312.

Losada, J.A., Montesinos, S., Omedas, M., Garcia, M.A. , Galvan, R., 2004. Cartography of the February-2003 flood at the Ebro river: Performance of photo interpretation, remote sensing and SIG anlysis in the GIS-EBRO. In Metodos Cuantitativos, SIG y Teledeteccion. (eds C. Conesa, Y. Alvarez , J.B. Martinez), Vol. 2, pp. 207-218. Universidad e Murcia, Murcia. (in Spanish).

Miller, J.R., Schulz, T.T., Hobbs, N.T., Wilson, K.R., Schrupp, D.L. , Baker, W.L., 1995. Changes in the landscape structure of a southeastern wyoming riparian zone following shifts in stream dynamics. . Biol. Conservation 72, 371-379.

McGarigal, K. , Marks, B., 1995. FRAGSTATS: Spatial Analysis Program for Quantifiying Landscape Structure. USDA For. Serv. Gen. Tech. Rep. PNW-351.

Ollero, A., 1992. The free meanders of the middle Ebro (Logroño-La Zaida): Fluvial geomorphology, eco-geography and risk analysis, Universidad de Zaragoza, Zaragoza. (in Spanish).

Ollero, A., 1995. Recent dynamics of the Ebro main channel at the Galachos natural reserve. Cuaternar. Geomorfol. 9, 85-93. (in Spanish).

Ollero, A., 2007. Channel adjustments, floodplain changes and riparian ecosystems of the middle Ebro River: Assessment and management. Int. J. Water Resour. Dev. 23, 73-90.

Pinilla, V., 2006. The development of irrigated agriculture in twentieth-century Spain: a case study of the Ebro basin. Agric. Hist. Rev 54, 122-141.

Poff, N.L., Allan, J.D., Bain, M.B., Karr, J.R., Prestegaard, K.L., Richter, B.D., et al.., 1997. The natural flow regime. Bioscience 47, 769-784.

Rohde, S., Hostmann, M., Peter, A. , Ewald, K.C., 2006. Room for rivers: An integrative search strategy for floodplain restoration. Landscape Urban Plann 78, 50-70.

Regato, P., 1988. Contribution to the study of flora and vegetation of La Alfranca oxbow lake and its relationship with the evolution of the fluvial system, Diputación General de Aragon, Zaragoza. (in Spanish).

Stanford, J.A., Lorang, M.S. , Hauer, F.R., 2005. The shifting habitat mosaic of river ecosystems. Ver. Int. Verein. Limnol. 29, 123-136.

Steiger, J., Gurnell, A.M., Ergenzinger, P. , Snelder, D., 2001. Sedimentation in the riparian zone of an incising river. Earth Surf Proc Land 26, 91-108.

Stromberg, J.C., Beauchamp, V.B., Dixon, M.D., Lite, S.J. , Paradzick, C., 2007. Importance of low-flow and high-flow characteristics to restoration of riparian vegetation along rivers in and south-western United States. Freshw. Biol. 52, 651-679.

Tockner, K., Malard, F. , Ward, J.V., 2000. An extension of the flood pulse concept. Hydrol. Process. 14, 2861-2883.

Tockner, K. , Stanford, J.A., 2002. Riverine flood plains: present state and future trends. Environ. Conser. 29, 308-330.

Vericat, D. , Batalla, R.J., 2006. Sediment transport in a large impounded river: The lower Ebro, NE Iberian Peninsula. Geomorphol. 79, 72-92.

Ward, J.V., Malard, F. , Tockner, K., 2002. Landscape ecology: a framework for integrating

pattern and process in river corridors. Landsc. Ecol. 17, 35-45.
Ward, J.V. , Stanford, J.A., 1995. Ecological Connectivity in Alluvial River Ecosystems and Its Disruption by Flow Regulation. Regul. Rivers Res. Manag. 11, 105-119.
Whited, D.C., Lorang, M.S., Harner, M.J., Hauer, F.R., Kimball, J.S. , Stanford, J.A., 2007. Climate, hydrologic disturbance, and succession: Drivers of floodplain pattern. Ecol. 88, 940-953.

STUDY II

The effect of anthropogenic disturbance on the hydrochemical characteristics of riparian wetlands at the Middle Ebro River (NE Spain)

The effect of anthropogenic disturbance on the hydrochemical characteristics of riparian wetlands at the Middle Ebro River (NE Spain)

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Abstract

In natural systems, the chemistry of floodplain waters is a function of the source of the water, which is influenced by geomorphic features of riparian wetlands. However, anthropogenic disturbances may alter both geomorphic features and the natural balance of water mixing in the floodplain. The aim of this study was to classify riparian wetlands and characterize their water characteristics in one reach of the Middle Ebro River to asses the hydrochemical functioning of the system. To accomplish that goal, water samples were collected at 40 sampling sites during low-water conditions and two floods of different magnitude. Moreover, geomorphic characteristics of riparian wetlands were also analyzed to interpret the results at broader spatio-temporal scales. Three group of wetlands were identified using multivariate ordination: (1) major and secondary channels highly connected to the river by surface water, containing weakly ionized water with high nitrate levels during floods; (2) secondary channels and artificial ponds located in riparian forests near the river, most of which were affected by river seepage during the examined events. This type of sites had high major ions concentrations and elevated spatial variability with respect to nutrient concentrations during floods. (3) Siltated oxbow lakes, whose hydrogeochemical features seemed to be unaffected by factors related to river fluctuations. Total dissolved solids, major ion (sulphate, chloride, sodium, calcium, magnesium, and potassium) and nutrient (nitrate, ammonium and organic nitrogen, and phosphate) depended upon the relationships between surface and subsurface water flows. Seasonal changes and geomorphic characterization indicated that a strong functional dependence of floodplain wetlands close to the main river channel is established, whereas most of the floodplain area remains disconnected from river dynamics. Moreover, the effect of nitrate-enriched agricultural runoff seems to affect water quality and hydrochemical gradients of the system. Based on our results, we propose different types of actions for the management of the Ebro River flow to ensure a more natural ecological functioning of its floodplains

Keywords: Hydrogeochemistry, floodplain, riparian wetlands, Ebro, alluvial aquifer, riverscape.

Introduction

To interpret the changes in ecological characteristics of riparian wetlands across multiple spatial and temporal scales, it is essential to understand the relationships between the river and its floodplain. Some have advocated using a four-dimensional (longitudinal, lateral, vertical and temporal) approach to describe fluvial hydrosystem and hydrological connectivity (Amoros et al., 1987; Ward 1989). Junk et al. (1989) emphasized the importance of viewing a river and its floodplain as a single dynamic system, where the driving force is the pulse of river discharge. On an annual or supra-annual scale, erosive floods are capable of creating and maintaining habitat patches at a variety of successional stages, which, in turn, determine the overall permeability and complexity of the landscape matrix (Metzger & Décamps, 1997; Galat et al., 1998; Geerling et al., 2006). On shorter time scales (days to months), the hydrological connectivity between landscape elements vary in response to the flow pulse, that is, water level fluctuations that are well below bankfull (Tockner & Stanford, 2002). Thus, hydrogeomorphic variables establish the physical template and the constraints under which chemical and biological processes can operate (Tabacchi et al., 1998).

Several studies have examined the effects of hydrological connectivity (superficial and river seepage inflow) on the water chemistry of riparian wetlands on multiple spatial scales (Trémolierés et al., 1993; Vandenbrink et al., 1993; Heiler et al., 1995; Knowlton & Jones 1997; Tockner et al., 1999; Malard et al., 2000; Domitrovic, 2003; Hein et al., 2004). Many have suggested that the chemistry of floodplain waters is a function of the source of the water. Thus, the influence of bank seepage from the main channel, vertical recharge during overbank flooding, the extent of the hyporheic corridor, the influence of local alluvial and hillslope aquifers, and inputs from main channel surface water can create a heterogeneous matrix of water bodies across the riverine landscape. The contribution of each water source is influenced by different geomorphic features of riparian wetlands (Amoros & Bornette, 2002). In addition, flow-paths and interactions between the various groundwater sub-compartments present considerable patchiness on different spatial and temporal scales (Brunke et al., 2003; McClain et al., 2003; Macpherson & Sophocleus, 2004; Lamontagne et al., 2005; Vallet et al., 2005).

However, anthropogenic alterations often disrupt the intensity, frequency, and timing of the natural disturbance regimes that maintain the ecological integrity of floodplain ecosystems (Ward & Stanford, 1995). The extent of riparian wetlands is decreasing. At the same time, wetland water quality and other ecological characteristics are often governed by factors unrelated to the interaction between hydrological connectivity and local environmental conditions. In addition, groundwater dynamics, including exchanges between groundwater sub-compartments, are affected by river regulation (Hancock, 2002). This, coupled with intensive agricultural use of floodplains, severely alters hydrodynamic and biogeochemical gradients (Hohensinner et al., 2004; Lamers, 2006).

In this study, we examined the hydrogeochemistry of riparian wetlands and characterized the geomorphological characteristics of the riverscape in one reach of the Middle Ebro River with the objectives of: a) identifying major hydrogeochemical processes affecting surface water physico-chemistry, b) defining wetland types with regards to their hydrochemical characteristics, taking into consideration the effect of floods, and c) discussing the results to provide a scientific support for ecological restoration after considering the hydrogeomorphological characteristics of riparian wetlands.

Methods

Study Area

The Ebro River, the largest river in Spain, is 910 km long, with a watershed of 85,362 km^2 and an average annual discharge into the Mediterranean Sea of 18138 Hm^3. The study area (Fig. 1) is in the Middle Ebro River, which remains geomorphologically active despite the presence of dams and reservoirs that have been built on the river and its tributaries. This section of the Ebro River is a meandering reach (sinuosity = 1.39, slope of the bankfull channel = 0.050%) with a floodplain that is, on average, about 5 km wide (Ollero, 1995). The average discharge at the Zaragoza gauging station (A011) located 12 km upstream of the study area is 223 $m^3 s^{-1}$ (1927-2003), and the elevation ranges from 175 m above sea level (asl) in the river channel to 185 m asl at the base of the scarp.

During the last century, the flow regime of the Ebro River has been greatly disrupted.

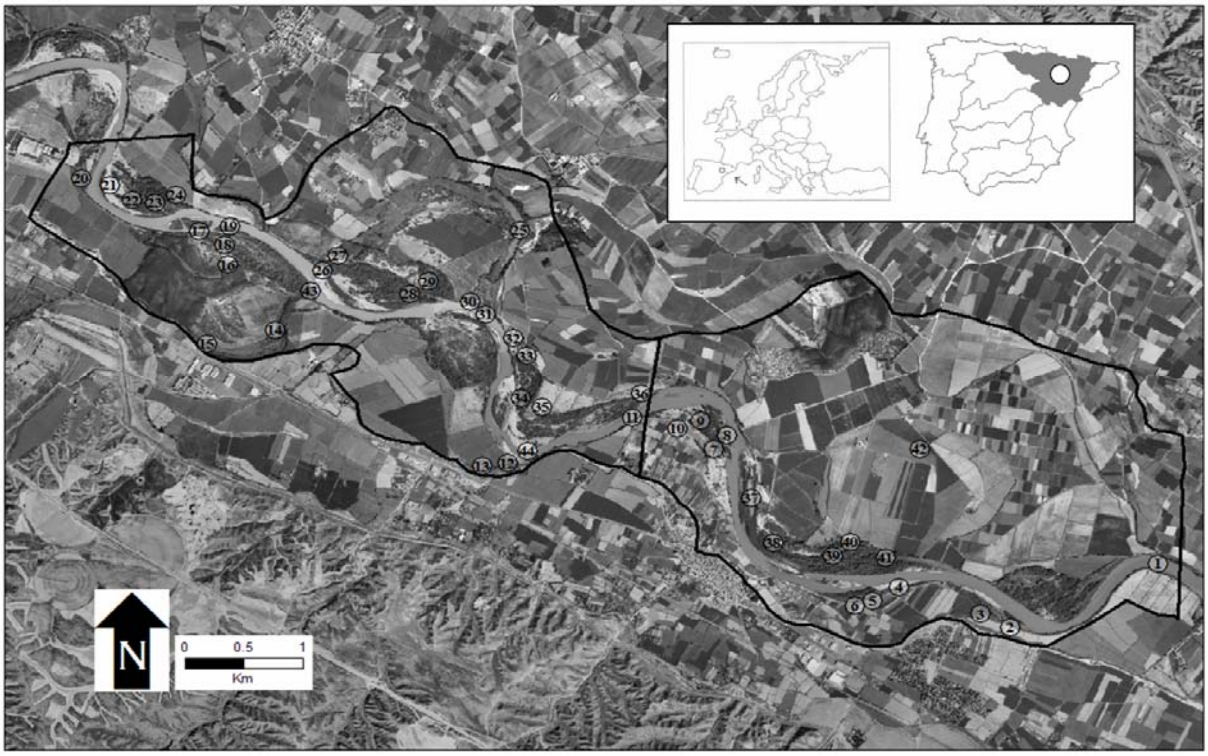

Figure 1. Locations of sampling sites within the study reach on the Middle Ebro River Floodplain, Spain. Dark line delimits the area flooded by a 3000 m^3 s-1 event (10-y return period, 1927-2003)

Irrigation of lowland areas and abandonment of farmland in upland areas has resulted in dramatic changes in both the hydrology, sediment load and river-floodplain interactions in the Ebro River (Ibañez et al., 1996; Batalla et al., 2004; Frutos et al., 2004; Pinilla, 2006; Ollero, 2007). The alluvial aquifer at the Middle Ebro is moderately contaminated by nitrate, presenting mean concentrations which ranges from 38 to 57 mg NO_3 L^{-1}, whereas concentrations in the main channel oscillates between 15 and 30 mg NO_3 L^{-1} (Torrecilla et al., 2005). Moreover, there is a strong seasonality in the aquifer yield to the Ebro River, peaking in summer as recharge from irrigation causes a general increase of the water table (FNE-UZ, 1993). At the study reach, average discharge has declined approximately 30% since 1981, whereas dykes have counteracted floodplain interactions since the 60´s (Ollero, 1992). Strong modification of the natural vegetation also ocurred (Regato, 1988; Castro et al., 2001), although, as far as we know, neither the effect of such human alterations over riparian wetlands has not been investigated, nor their hydrochemical features have been described.

Hydrogeomorphological setting

All river discharge values used in this paper are daily averages discharges measured at the Zaragoza gauging station (A011), located 12 km upstream of the study area. This information, as well as values for water table height, was provided by the Ebro River Basin Administration (www.chebro.es). Recurrence intervals for different magnitude floods, including bankfull discharge (1.52 y), were estimated from daily average discharge values amongst 1927 to 2003, using the General Pareto distribution as described by Beguería (2005). The characterization of alluvial aquifer fluctuations was based on data from one well (piezometric control point) located 8 km downstream of the study area and 1.3 km from the main channel.

Water table heights were recorded at least every two months from 2002 to 2006.

To interpret the main features and wetland types obtained by the hydrogeochemical analysis within a study reach perspective, a geomorphological characterization of the riverscape (sensu Malard et al., 2002) was performed. Field-based information and ground verification of 1998 aerial photographs (1:15000) of the study area were used to delimit riparian wetlands (hereafter, RWs), defined as permanent water bodies and their surrounding areas. Wetland limits were defined by human landscape features (e.g., paths and crops) for the oxbow lakes and artificial ponds. For the outer banks of side channels and backflow channels, limits were demarcated by abrupt topographic changes. The limits of sides adjacent to the riverbank were marked by the point bars associated with the main channel. For spatial calculations, the total area of the floodplain was estimated using information from the Ebro River Basin Administration, which used remote sensing techniques (Losada et al., 2004) to map the flooded area during the February 2003 flood (10-y return period event). The map was digitized and area was calculated with ArcView® 8.3.

To facilitate the identification of wetland types, the potential influence of alluvial aquifer was evaluated using total dissolved solids (TDS) and nitrate concentrations . Due to the evaporitic nature ($CaSO_4$ and NaCl) of the substrate at the study reach, field-measured conductivity, which is directly related to TDS, had been used previously for qualitative estimations (Torrecilla et al. 2002). To asses the current validity of these variables as qualitative indicators, we characterized the alluvial aquifer using data from the nearest well located 4 km upstream of the study reach and 1.8 km from the main channel (n = 20, 1997-2005), provided by the Ebro Basin administration (www.chebro.es).

Sampling Protocols

To explore the effect of river fluctuations on RWs water quality, water was sampled during different seasons. Water samples were collected once per season in the spring (38 sites), summer (24 sites), and winter (34 sites) of 2004. Every effort was made to collect samples from all water bodies in the study area during each sampling period, including one irrigation channel, in order to ensure the greatest variety of RWs with respect to hydrological connectivity (Fig. 1). Spring samples were collected during a 720 $m^3\ s^{-1}$ flood (0.21 y return period), which followed other six flood events, all below the bankfull discharge (1450 $m^3\ s^{-1}$, 1.52 y return time). Summer samples were collected only at those sites that retained water after a long period of low river flow (< 50 $m^3\ s^{-1}$). Winter samples were collected during a 423 $m^3\ s^{-1}$ flood (0.14 y return time).

Surface-water samples (at a 10-cm depth) were collected directly into 1.5 L acid-washed PVC bottles and transported in dark cool-boxes to the laboratory. A roped bucket was used to take water samples so as to minimize bank proximity effects. Upon arrival in the laboratory, total suspended solids (TSS) was determined by filtering samples through pre-combusted (450 ºC, 2 h) Whatman© GF/F glass-fiber filters, followed by drying filters at 60ºC until a constant weight was reached. Alkalinity of unfiltered water was estimated within 4 h of collection by automatic titration with H_2SO_4 0.04 N (APHA, 1989). Total dissolved solids (TDS) were determined by weighing the residue after evaporating 25 ml of filtered water at 100ºC (APHA, 1989). Aliquots of filtered water were stored at –20ºC and later used for remain analyses within one month. Total nitrogen (TN) and other dissolved inorganic forms of nitrogen (DIN=nitrate [$N\text{-}NO_3^-$], nitrite [$N\text{-}NO_2^-$], ammonium [$N\text{-}NH_3$]) and orthophosphate ($P\text{-}PO_4^{3-}$), were quantified colorimetrically using a semiautomatic flow analyser (SYSTEA©) following standard protocols (APHA 1989). Dissolved organic nitrogen (N-org) was calculated as the difference between total nitrogen and the nitrogen content of all species of dissolved inorganic nitrogen. The major dissolved ions, sulphate ($SO_4^{=}$), chloride (Cl^-), calcium (Ca^{++}), magnesium (Mg^{++}), sodium (Na^+), and potassium (K^+), were quantified using high performance liquid chromatography (APHA, 1989). Conductivity, temperature, pH, and dissolved oxygen were measured *in situ* using multiple probes.

To determine differences caused by different hydrological conditions data sets were individually subjected to multivariate statistical analyses involving Principal Component Analysis (PCA) and Cluster analysis based on a Ward algorithm (SPSS© 14.0 package). Data were transformed to satisfy conditions of normality, when necessary. In spring, the analysis was performed separately for wetlands located at the left or right bank. When including all spring sites in the same analyses, the hydrogeochemical interpretation of factors identified was unclear because some of the variables presented high factorial loadings for different factors. Differences among clusters were tested using the non-parametric Kruskal-Wallis Test for water physico-chemical variables. PCA factors and the physico-chemical features of cluster groups were used to identify the main hydrogeochemical processes. Afterwards, wetlands types were defined using the cluster solutions for the three examined periods. To characterize the examined wetlands, PCA scores and chemical indicators were represented by grouping sites according to wetland type.

Results

Hydrogeomorphological setting

Riparian wetlands (RWs) occupied 3.86% of the entire floodplain area, but only 0.69% of the considered area was permanently flooded (Tab. 1). Oxbow lakes covered most of the area occupied by RWs although the permanently flooded area of these wetlands was low (10%). Backflow and side channels covered scant portions of the floodplain area and also accreted areas dominated over flooded zones. With regards to the alluvial aquifer, Conductivity (1722.72 ± 190.37 µS cm^{-1}) and nitrate concentrations were consistently high (60.23 ± 12.32 mg L^{-1}). Despite the low frequency of observations (n = 20, 1997-2005), the data set was considered representative of the variables under consideration because of its low variability. Aquifer water table exhibited consistent seasonal pattern of fluctuations over several years, with peak levels in September, when river discharges were the lowest (Fig. 2).

Hydrochemical characteristics

From the cluster analysis of hydrochemical data (or water chemistry data) we obtained several groups of sampling sites (Fig. 3), which differed for geomorphic features (Tab.2) and water quality (Tab. 3 and 4). In Spring, wetlands in the LOWTDS group, which included main channel locations at both margins, had higher levels of suspended solids, nitrate, oxygen and phosphate if compared with HIGHTDS. They also exhibited relatively high pH values, lower levels of both TDS and major ions (Tab. 3). HIGHNIT sites exhibited the highest TDS levels and the highest concentrations of oxygen, chloride, sodium, nitrate and organic nitrogen. All sites that belongs to the HIGHNIT group were located in the left margin, namely the irrigation channel, an ox-bow lake, and two old gravel pits close to the latter oxbow lake.

In summer, the affinity between LOWTDS and HIGHTDS decreased (Fig. 3). The water in the HIGHTDS wetlands was slightly more oxygenated and diluted, with lower concentrations of all nitrogen species than the LOWTDS wetlands. LOWTDS included

RW Type	AREA		PFA		
	Total (Ha)	% floodplain	Total (Ha)	% floodplain	% RW
Artificial ponds	3.35	0.15	2.34	0.11	69.85
H.C Side channels	6.88	0.31	2.21	0.10	32.12
Oxbow lakes	49.95	2.24	5.03	0.23	10.07
Backflow channels	11.88	0.53	3.57	0.16	30.05
Side channels	13.99	0.63	2.15	0.10	15.37
	86.03	3.86	15.31	0.69	100
Main channel	125.46	5.63	125.46	5.63	
Total	211.50	9.48	140.78	6.31	

Table 1. Extent of Riparian Wetlands on the floodplain under study, delimited by the area flooded in a 25-year event. Areas covered by each different geomorphological type are shown. RW = Riparian Wetland; PFA= Permanently Flooded Area; H.C= Highly Connected.

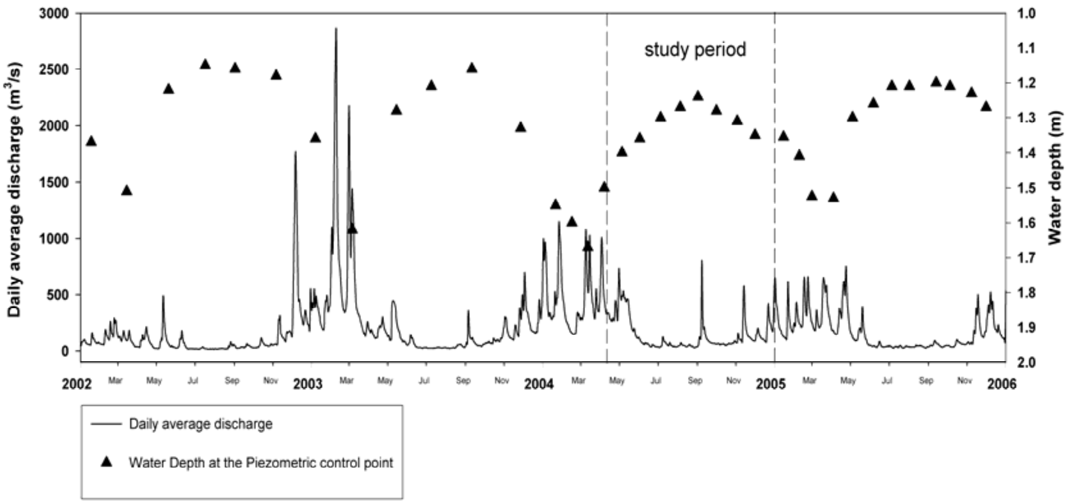

Figure 2. Depth of the alluvial aquifer at the piezometric control point and average daily discharge of the Ebro River, Spain, at the Zaragoza gauging station (2002-2006).

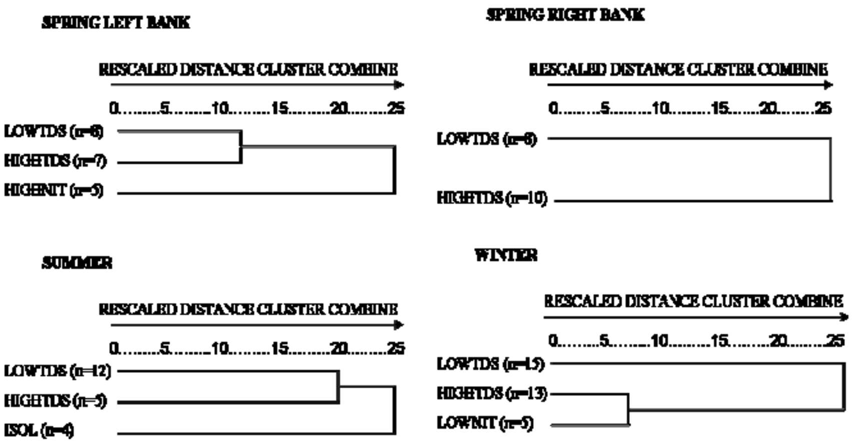

Figure 3. Location dendrograms generated by various cluster analyses. For ease of interpretation, sample sites are grouped by cluster results

Site	Description	Cluster Groups			Wetland Type	Site	Description	Cluster Groups			Wetland Type
		Spring	Summer	Winter				Spring	Summer	Winter	
1	Main channel	LOWTDS	LOWTDS	LOWTDS	**HIGHCON**	23	Backflow channel	---	----	HIGHTDS	**INTCON**
2	Side channel	LOWTDS	LOWTDS	LOWTDS	**HIGHCON**	24	Backflow channel	LOWTDS	LOWTDS	HIGHTDS	**INTCON**
3	Backflow channel	HIGHTDS	ISOL	HIGHTDS	**INTCON**	25	Oxbow lake	HIGHNIT	HIGHTDS	HIGHTDS	**LOWCON**
4	Backflow channel	LOWTDS	LOWTDS	LOWTDS	**HIGHCON**	26	Point Bar	LOWTDS	LOWTDS	HIGHTDS	**INTCON**
5	Side channel	LOWTDS	LOWTDS	LOWTDS	**HIGHCON**	27	Artificial Pond	HIGHNIT	+	+	**INTCON**
6	Side channel	---	---	HIGHTDS	**INTCON**	28	Side channel	LOWTDS	HIGHTDS	LOWTDS	**HIGHCON**
7	Side channel	LOWTDS	+	LOWTDS	**HIGHCON**	29	Artificial Pond	HIGHNIT	+	+	**INTCON**
8	Side channel	LOWTDS	LOWTDS	LOWNIT	**INTCON**	30	Oxbow lake	HIGHNIT	ISOL	HIGHTDS	**LOWCON**
9	Riparian Forest	HIGHTDS	+	+	**INTCON**	31	Main Channel	LOWTDS	LOWTDS	LOWTDS	**HIGHCON**
10	Artificial Pond	HIGHTDS	HIGHTDS	LOWNIT	**INTCON**	32	Artificial Pond	LOWTDS	+	+	**INTCON**
11	Main channel	LOWTDS	LOWTDS	LOWTDS	**HIGHCON**	33	Artificial Pond	HIGHTDS	+	+	**INTCON**
12	Oxbow lake	HIGHTDS	+	+	**LOWCON**	34	Side channel	LOWTDS	HIGHTDS	HIGHTDS	**INTCON**
13	Oxbow lake	HIGHTDS	+	+	**LOWCON**	35	Artificial Pond	HIGHTDS	+	+	**INTCON**
14	Oxbow lake	HIGHTDS	ISOL	HIGHTDS	**LOWCON**	36	Backflow channel	LOWTDS	+	LOWTDS	**HIGHCON**
15	Oxbow lake	HIGHTDS	+	+	**LOWCON**	37	Side channel	HIGHTDS	LOWTDS	HIGHTDS	**INTCON**
16	Side channel	HIGHTDS	+	LOWNIT	**INTCON**	38	Riparian Forest	---	---	HIGHTDS	**INTCON**
17	Main channel	LOWTDS	LOWTDS	---	**HIGHCON**	39	Riparian Forest	HIGHTDS	+	LOWTDS	**INTCON**
18	Riparian Forest	HIGHTDS	+	LOWNIT	**INTCON**	40	Artificial Pond	HIGHTDS	+	+	**INTCON**
19	Side Channel	---	---	LOWNIT	**INTCON**	41	Backflow channel	HIGHTDS	LOWTDS	HIGHTDS	**INTCON**
20	Side channel	HIGHTDS	+	HIGHTDS	**INTCON**	42	Irrigation Channel	HIGHNIT	HIGHTDS	HIGHTDS	**INTCON**
21	Point Bar	LOWTDS	+	LOWTDS	**HIGHCON**	43	Backflow channel	---	---	LOWTDS	**HIGHCON**
22	Riparian Forest	HIGHTDS	+	HIGHTDS	**INTCON**	44	Side channel	---	ISOL	LOWTDS	**HIGHCON**

Table 2. Classification of sampling sites by wetland types for the three examined periods using cluster analysis (see results). "---" = Not Sampled; "+"= No Water. ; HIGHCON: Low mineralized wetlands; INTCON: Intermediate mineralized Wetlands; LOWCON: High mineralized wetlands.

	LEFT BANK					RIGHT BANK			
	MEAN ±S.E			STATISTICS		MEAN ±S.E		STATISTICS	
	HIGHNIT	LOWTDS	HIGHTDS	X^2	P-value	LOWTDS	HIGHTDS	X^2	P-value
pH	7.55±0.15	8.09±0.09	7.85±0.11	8.40	0.02*	8.09±0.07	7.63±0.11	6.88	0.01*
O2 (mg L^{-1})	8.06±0.67	7.37±0.40	6.97±0.63	0.57	0.75	8.27±0.42	6.88±1.02	2.02	0.15
O2 (%)	82.00±8.65	71.44±3.90	69.63±6.97	0.57	0.75	82.38±4.53	68.87±10.68	2.02	0.16
TSS (mg L^{-1})	13.65±6.40	109.21±20.52	7.93±2.09	13.47	0.00**	32.96±11.59	16.77±4.10	0.64	0.42
TDS (mg L^{-1})	1409.11±219.19	516.14±12.38	742.86±66.77	13.68	0.00**	462.00±27.27	1010.80±125.41	12.64	0.00**
Cl (mg L^{-1})	288.03±89.72	82.95±4.04	121.73±10.63	13.67	0.00**	58.15±4.84	109.80±20.75	8.08	0.00**
SO4 (mg L^{-1})	303.05±80.26	136.86±6.22	164.70±6.93	11.40	0.00**	89.23±7.41	296.07±74.13	10.23	0.00**
Na (mg L^{-1})	160.06±21.83	41.70±3.75	58.95±6.87	12.44	0.00**	52.60±1.49	100.28±16.57	12.63	0.00**
K (mg L^{-1})	3.38±0.47	3.42±0.62	5.96±1.26	3.34	0.19	2.71±0.27	5.08±0.63	9.67	0.00**
Ca (mg L^{-1})	102.25±3.33	50.22±4.70	61.00±3.92	12.21	0.00**	68.78±3.08	140.59±23.39	8.60	0.00**
Mg (mg L^{-1})	23.05±1.80	11.85±1.24	11.96±1.59	9.60	0.01**	14.83±0.49	22.22±2.21	12.01	0.00**
NO3 (mg NO3 L^{-1})	7.04±2.15	4.40±0.26	1.50±0.57	9.38	0.01**	5.16±0.33	2.61±0.75	7.11	0.01**
PO4 (mg PO4 L^{-1})	0.18±0.03	0.24±0.03	0.27±0.04	3.21	0.20	0.24±0.02	0.13±0.02	6.64	0.01**
Norg (mg-N L^{-1})	1.70±0.71	1.53±0.10	0.50±0.15	8.07	0.02*	1.18±0.12	1.45±0.68	1.14	0.29
NH4 (mg NH4 L^{-1})	0.24±0.16	0.07±0.03	0.04±0.01	2.90	0.23	0.09±0.02	0.45±0.43	3.48	0.06

Table 3. Mean values and standard error for water chemistry variables included in the multivariate analysis for spring. Sites have been grouped using results of cluster analysis. Results of Kruswall-Wallis non-parametric test are shown; X^2 = squared chi; *p < 0.05; **p < 0.01.

	SUMMER					WINTER				
	MEAN ±SE			STATISTICS		MEAN ±SE			STATISTICS	
	LOWTDS	ISOL	HIGHTDS	X^2	P-value	LOWTDS	HIGHTDS	LOWNIT	X^2	P-value
pH	7.59±0.06	7.54±0.06	7.62±0.12	0.40	0.82	7.90±0.07	7.70±0.09	7.72±0.17	3.44	0.18
O2 (mg L^{-1})	5.37±0.31	9.47±1.00	6.26±0.56	9.87	0.01**	8.30±0.56	7.92±0.63	7.17±1.81	0.63	0.73
O2 (%)	63.28±3.95	105.98±9.53	71.88±5.90	9.63	0.01**	67.74±4.53	66.28±4.89	56.80±14.18	0.69	0.71
TSS (mg L^{-1})	26.43±3.67	23.06±5.98	17.51±5.93	0.94	0.62	41.20±8.36	14.29±2.26	10.54±2.87	14.24	0.00**
TDS (mg L^{-1})	1465.67±71.53	2253.00±235.13	1270.40±63.49	11.33	0.00**	584.60±35.30	1540.31±213.46	1484.00±178.61	23.18	0.00**
HCO_3 (mg L^{-1})	268.33±5.55	225.55±18.31	181.26±13.78	12.88	0.00**	273.37±6.76	313.37±13.87	378.40±17.94	12.16	0.00**
Cl (mg L^{-1})	388.58±11.56	421.67±73.89	375.40±28.58	0.18	0.92	115.19±13.73	339.17±42.28	359.71±45.47	22.38	0.00**
SO4 (mg L^{-1})	589.19±44.82	1129.96±198.37	467.31±18.78	12.31	0.00**	165.03±14.51	574.31±65.76	428.12±33.33	24.22	0.00**
Na (mg L^{-1})	203.85±13.55	233.60±39.48	192.59±14.02	1.18	0.55	71.58±7.41	202.66±29.16	199.88±27.14	20.63	0.00**
K (mg L^{-1})	5.15±0.30	7.25±1.43	7.13±1.33	3.96	0.14	3.16±0.31	6.74±0.61	4.92±1.15	18.94	0.00**
Ca (mg L^{-1})	153.85±12.88	282.92±52.13	131.14±4.29	9.57	0.01**	67.66±4.42	155.28±17.49	140.82±14.25	23.13	0.00**
Mg (mg L^{-1})	37.22±1.32	46.79±2.04	29.08±1.80	11.88	0.00**	13.62±1.12	33.11±3.81	26.94±1.96	23.00	0.00**
NO3 (mg NO3 L^{-1})	14.47±1.18	15.85±6.39	8.23±3.12	3.90	0.14	4.95±0.38	4.41±0.91	0.29±0.14	12.55	0.00**
PO4 (mg PO4 L^{-1})	0.20±0.01	0.20±0.02	0.18±0.01	0.99	0.61	0.40±0.13	0.28±0.05	0.10±0.06	3.85	0.15
Norg (mg-N L^{-1})	2.71±0.70	2.14±1.03	0.80±0.32	4.42	0.11	2.49±0.16	3.31±0.68	0.34±0.03	12.13	0.00**
NH4 (mg NH4 L^{-1})	0.35±0.07	0.08±0.03	0.06±0.02	13.51	0.00**	0.07±0.02	0.07±0.02	0.02±0.01	6.02	0.05*

Table 4. Means ± standard errors for water chemistry variables included in the multivariate analysis for summer and winter. Sites have been grouped using results of cluster analysis. Results of Kruswall-Wallis non-parametric test are shown. X2 = squared chi; *p < 0.05; **p < 0.01.

locations at the river and adjacent side and backflow channels (Tab. 2) which presented higher concentrations of suspended solids (Tab. 4) than HIGHTDS. A group of locations with isolated waters that included sites at oxbow lakes, one side and one backflow channel (Tab. 2) was differentiated and labelled as isolated locations (ISOL). ISOL sites also exhibited high suspended-solids concentrations, high oxygen concentrations and near-outlier scores for dissolved solids and associated variables.

In winter, HIGHTDS and LOWNIT groups were clustered with high affinity (Fig. 3). The LOWNIT group included one artificial pond, one riparian area and side-channels (Tab. 2), all located at the right margin. In turn, most of the sites in the HIGHTDS group were on the left bank, including the irrigation channel (Fig. 1). The HIGHTDS and LOWNIT sites had higher ion concentration than LOWTDS sites, which included main channel locations. Nutrient levels at HIGHTDS sites were similar to those of the LOWTDS sites (Tab. 4). LOWNIT presented lower concentrations of nitrate, phosphate and organic nitrogen, whereas ammonia was similar for all groups.

Season	Bank	Factor	Explained variance	Accumulated variance	Related Variables
Spring	**Left**	1	37.17	37.17	Ca^{++} ,$SO_4^{=}$,Na^{+} ,TDS, Cl^{-}
		2	19.7	56.87	O_2 % ,O_2
		3	12.05	68.92	K^{+} ,NH_4^{+} , Norg
		4	11.38	80.3	TSS, NO_3^{-}
Spring	**Right**	1	35.78	35.78	$SO_4^{=}$, Na^{+} ,Ca^{++} ,K^{+} ,Mg^{++} ,$PO_4^{=}$,TDS
		2	20.12	55.9	pH ,O_2 % ,O_2
		3	15.57	71.47	NO_3^{-} ,$NH4^{+}$,Norg
Summer	**Both**	1	22.99	22.99	Mg^{++} ,TDS ,Ca^{++} ,$SO_4^{=}$
		2	16.97	39.96	$NH4^{+}$,O_2 ,O_2 %
		3	16.16	56.12	HCO_3^{-} ,NO_3^{-} ,Norg.
Winter	**Both**	1	39.18	39.18	Ca^{++} ,Na^{+} ,Cl^{-} ,$SO_4^{=}$,TDS ,Mg^{++}
		2	18.74	57.93	$NH4^{+}$,NO_3^{-} ,Norg
		3	17.5	75.43	pH ,O_2 % ,O_2

Table 5. PCA solutions for the analyses performed. For ease of interpretation, only variables with higher factor loadings (> 0.65) are included.

The PCA results are reported in Tab. 5. The first two factors explained approximately 50 % of the total variance. The first PCA factor was related to TDS and major ions ($SO_4^{=}$, Cl^{-}, Na^{+}, Ca^{++}, Mg^{++}, K), although the proportion of the variance explained by this factor decreased in summer. Variables associated with this factor and their loadings changed over time, and only sulphate and calcium exhibited high scores for this factor during all periods examined. The second and third principal components accounted for between 16% and 20% of the total variance in the data. Dissolved oxygen reached high factorial loadings for the second component in spring and summer. In contrast, nutrient concentrations (NO_3^{-}, NH_4^{+}, $PO_4^{=}$, N-org) were highly correlated with the third component in spring and winter, but with the second component in summer.

Wetland types

Three RWs were identified: 1) High connected (HIGHCON, n = 13): Sites classified as LOWTDS in spring and winter; those sites included in the LOWTDS cluster in winter but not sampled in spring were also classified as this type (Tab. 2). HIGHCON locations presented the most diluted waters in winter and spring (Fig. 4). The scores for the PCA factor associated with nutrients were markedly higher than those of the other groups for that season. Nitrate and TDS values for HIGHCON locations were two-fold higher in summer than during spring or winter (Fig. 5). Nitrate concentrations for HIGHCON locations were also higher than those at other locations in spring and summer.

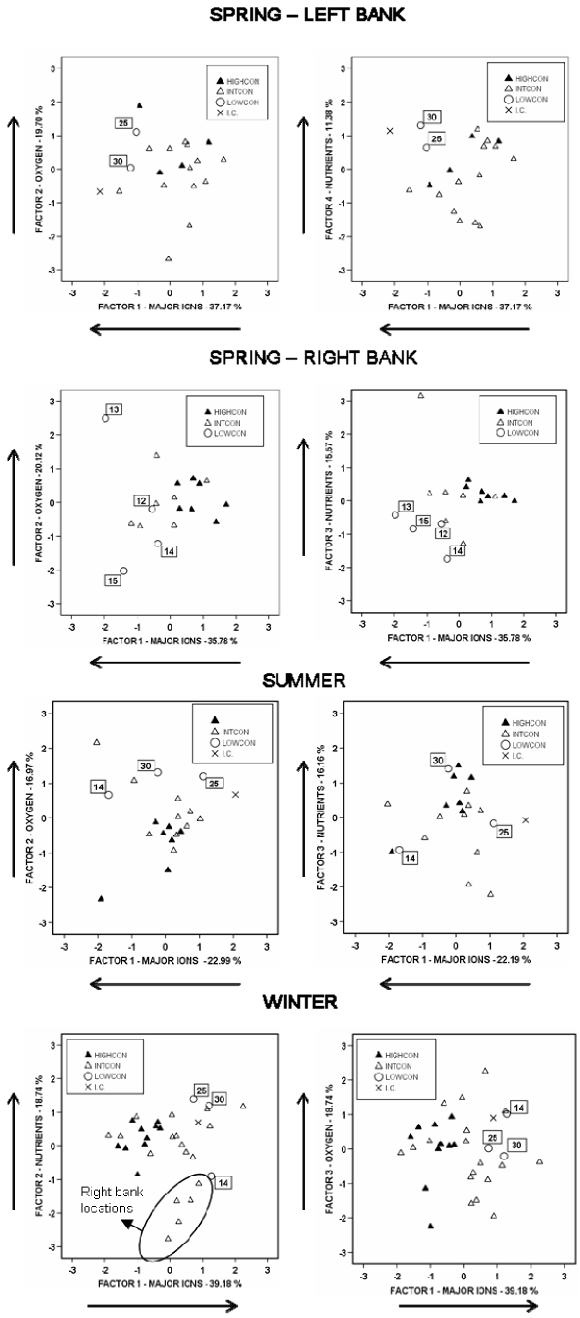

Figure 4. Position of sample sites on the factors extracted by the various PCA analyses. Arrows indicate an incremental gradient of factor loading. Locations have been grouped by wetland type (see results): HIGHCON: High connected ; INTCON: Intermediate connected ; LOWCON: Low connected; I.C: Irrigation Channel. LOWCON sites are highlighted.

2) Intermediate connected with the main channel (INTCON, n = 19): Heterogeneous group of locations that were not classified as LOWTDS in winter (Tab. 2). In winter, at most of these sites, TDS and ion concentrations attained higher concentrations that in HIGHCON sites; this was also the case in spring for right-bank sites that were also in this group (Fig. 4). Scores for the nutrients PCA factor in summer were slightly lower, and these differences increased in winter for a group of sites located at the right bank. At INTCON sites, TDS and nitrate concentrations were markedly higher in summer than in other seasons (Fig. 5). In summer, TDS values did not differ from the HIGHCON sites as substantially as they did in spring and winter, although differences in winter were somewhat more marked. Nitrate concentrations were highly variable in spring and winter, with a group of sites at the right margin showing negligible concentrations of this nutrient.

3) Low connected (LOWCON, n = 5): Sites not classified as LOWTDS in any of the seasons (Tab. 2). They exhibited the highest ionic concentrations in spring and winter. Only site 14, located at the right margin, differed markedly from the rest of the sites in summer (Fig. 4). Sites 25 and 30, located at the left margin, had elevated scores for nutrient and oxygen factors in all seasons. Site 14 exhibited low oxygen values in summer, and nutrient factor scores that fell below the mean in all seasons, as sites 12, 13, and 15, which were temporary or which

were submersed only in spring. TDS was elevated at each of the five locations. In all cases, summer and winter values were slightly higher than spring values (Fig. 5)
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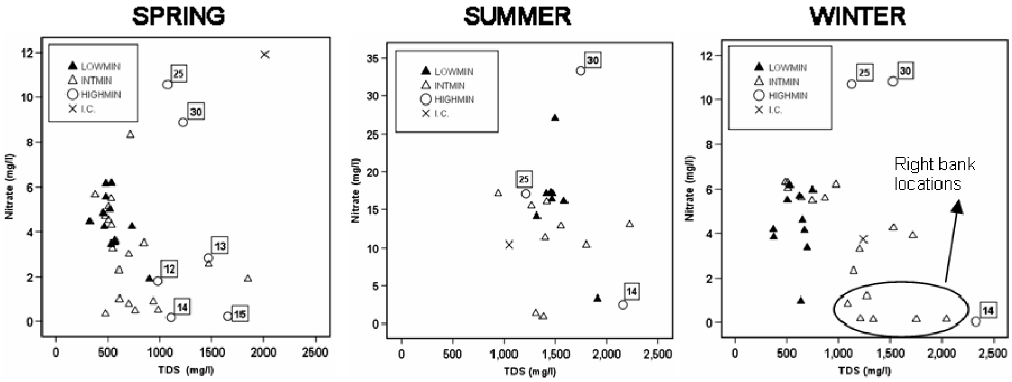

Figure 5. Bivariate diagrams for the geochemical indicators, Total Dissolved Solids (TDS) and Nitrate. Locations have been grouped by wetland type (see results): HIGHCON: High connected; INTCON: Intermediate connected; LOWCON: Low connected; I.C: Irrigation Channel. LOWCON sites are highlighted

Nitrate concentrations were almost negligible at site 14, in the right margin, whilst nitrate peaked in summer in sites 25 and 30.

Discussion

Hydrochemical characteristics

Our study evidenced that ionic concentrations resulted from the ratio of water flow from the river and groundwater discharge. The balance of surface and groundwater has been already reported as the main factor in determining the changes in water quality in the riverscape during flood events of varying magnitudes (Malard et al., 2000; Arntzen et al., 2006; Malard et al., 2006). In spring and winter, main channel inputs of low conductivity and turbid waters differentiated the water chemistry of superficially connected wetlands, which were clustered with river locations. Conversely, lower suspended solids content and more ion-rich waters characterized the other groups in spring and winter, possibly reflecting the relative dominance of groundwater inputs. Most of these locations were located adjacent to the main channel and were permanent water bodies also in summer. We also found that, with the exception of oxbow lakes, the hydroperiods of these environments were related with river discharge fluctuations. Likely, water quality was influenced by hyporheic flows and our results might reflect the balance of river seepage and inputs from a ion rich aquifer. According to our

characterization, the high conductivity in the water of the alluvial aquifer water is persistent through time, reflecting the strong influence of the evaporitic substrate (NaCl and CaSO4) in the groundwater of the studied reach. In spring, following several recurrent floods, the differences between LOWTDS and HIGHTDS were lower than they were in winter, when samples were taken during the lowest magnitude flood. This might be explained by a greater influence of lower salinity river seepage relative to that of the alluvial aquifer in spring. In summertime, the alluvial water table was at its highest levels and river discharge was at its lowest levels (Fig. 2); these factors appear to influence both main channel and floodplain waters, as shown by their similar and high major ions concentrations.

A second major forcing function was associated mainly with nitrogen species as a result of nutrient delivery from the main river into the adjacent floodplain during flood-pulses (Van der Brink et al., 1993; Heiler et al., 1995; Hein et al., 1999; Tockner et al., 1999). Clusters that included main channel locations exhibited the highest nutrient concentrations for almost all examined seasons. The spatial and temporal heterogeneity in the remainder groups could be explained by within site differences depending either on nutrient processing or on water inputs. For example, a group of sites located in the right bank presented low nitrate contents over the all examined periods (Fig. 5). It could be explained by the fact that the irrigated area theoretically draining to the main channel is much larger at the left bank (Fig. 1), presumably affecting the buffer capacity of riparian areas at both margins to ameliorate the nitrate-enriched agricultural runoff. The separation of HIGHNIT sites, all located at the left margin and characterized by higher nitrogen content, supports this idea. Sanchez-Perez et al. (2003) stressed the importance of the impact of nitrate rich groundwater over water at the main channel and the riparian area at the Garonne River (France). Such dominance of euthrophic groundwater could occur at the study reach as a result of irrigation recharge, especially in summer when the aquifer yield to the Ebro River is at its highest (FNE-UZ, 1993). This dynamic might also be responsible for promoting the highest nitrate concentrations (30 mg L^{-1}) at the main channel in August (Torrecilla et al., 2005).

The effect of the Waste Water Treatment station of Zaragoza city was initially considered, the effluent being an important source of nitrate to the river. However, this hypothesis was later discarded because the effect over river water disappeared 500 m downstream (unpublished data).

With regards to the oxygen, no clear trends in dissolved oxygen for the examined sites in spring or winter, whereas in summer, the group that included main channel locations showed only slightly higher oxygen concentrations. A detailed metabolic characterization of the study sites is required to explain the data variability in terms of oxygen concentration.

Wetland types

On the basis of surface water chemistry during all examined periods, three types of wetlands were identified and associated with possible water sources: 1) HIGHCON. This type comprises river locations, as well as side and backflow channels and flooded point bars located adjacent to the main channel. Superficial inputs from the main channel determined the water chemistry at these sites, even during high frequency events (400 $m^3 s^{-1}$; 0.12 y). 2) INTCON. This is a heterogeneous group of sites that includes side and backflow channel locations, artificial ponds and flooded riparian areas. They share that inputs from the main channel, either superficial or seepage, dominates during an 800 $m^3 s^{-1}$ event (0.23 y return time) and after several higher magnitude floods. The water chemistry of these sites did not appear to be as influenced by superficial inputs from the main channel during the winter flood as were HIGHCON locations. 3) LOWCON. This type comprises oxbow lakes positioned far from the river. Inputs from the main channel did not appear to control surface water chemistry at these sites to the same extent as they did for the other wetland types. Instead, alluvial inputs seemed to predominate. Moreover, thick layers of bottom sediment probably counteracted subsurface water exchanges, suggesting that *in situ* stagnation processes carved the hydrogeochemical features of this type. Previous studies have reported similar differences between study sites, reflecting the importance of the hydrological connectivity on water chemistry (Knowlton and Jones, 1997; Bornette et al., 1998; Hein et al., 2004), what can change over time for a given site (Tockner et al., 1999; Malard et al., 2000, Thomaz et al., 2007).

At hydrologically connected sites (i.e., HIGHCON and INTCON), low water levels decreased heterogeneity on major ions concentration (Fig. 4 and 5). Divergences also diminished during a relatively high frequency flood (0.23 y), especially at the left bank, paced by slightly larger magnitude events (0.37 y). However, inputs of superficial water during the lowest magnitude flood examined primarily altered the most-connected HIGHCON locations, creating the widest range in mineralization conditions. With respect to nutrients, hydrological conditions in spring appeared to favour nitrate depletion independently of site location; but in winter, this effect was observed only for a group of sites at the right bank (Fig. 4 and 5). Further research is required to delimitate the existence of "hot spots" and "hot moments", which is common at the interface of aquatic and terrestrial ecosystems as noted by McClain et al. (2003). In summer, after six months of intensive irrigation, HIGHCON and INTCON sites appeared to be greatly affected by a nutrient-rich agricultural runoff that tended to homogenize differences.

At disconnected oxbow lakes, the position in the landscape could explain differences in water chemistry. The waters of site 14 (located at the right margin) were more saline and nitrate-poor than waters at sites 25 and 30, located at the left margin (Fig. 1). Water stagnation might explain the hydrogeochemical features of site 14, which had a thick layer of fine sediment that probably counteracted vertical water exchange. However, during low frequency floods (2000 m^3 s^{-1}; 1.95 y), superficial inputs have been observed to occur. In turn, the irrigated area theoretically draining to the main channel is much larger at the left bank, as described in the previous section (Fig. 1). Thus, higher alluvial aquifer pressure over sites 25 and 30 could influence water chemistry as previously reported by Torrecilla et al. (2002). The close position of the irrigation channel to site 25 in the TDS/NO_3 plot (Fig. 5) during periods of active irrigation (i.e., spring and summer) supports this idea.

Hydrogeomorphological constraints and ecological restoration

The examined wetlands exhibited a wide range of hydrogeochemical features during the examined floods. Because almost all water bodies were sampled at each period, these features are highly representative of the entire reach. However, the observed heterogeneity was limited to the scarcely flooded areas of the RWs (Tab. 1). Flow regulation for agricultural purposes and dike construction for flood protection have largely modified river-floodplain interactions during the last century at the study reach (Castro et al., 2001; Ollero, 2007; Cabezas et al., in press). The absence of channel migration since 1981 (Ollero, 1995) has probably caused vertical accretion to dominate floodplain formation. As a result, the majority of the floodplain area is flooded only during high magnitude floods (> 2500 m^3 s^{-1}, 5.36 y return period) and remains disconnected from the river most of the time.

In addition, floods may increase the similarity among aquatic habitats in our study area, As pointed out by Thomaz et al., (2007). Side and backflow channels accounted for a very small portion of the floodplain and were highly accreted (Tab. 1), suggesting that potentially flooded areas are highly terrestrialized in the study reach. At HIGHCON and INTCON sites, which were the most common wetland types in these areas, we hypothesize that water sources will rapidly become dominated by superficial inputs during events larger than those considered in this study. In turn, the area covered by oxbow lakes is relatively bigger and dominates over other wetland types, although the portion of permanently flooded area is also low. LOWCON sites were located at these wetlands, where water exchange with groundwater was counteracted by thick layers of bottom sediment. Superficial inputs to these sites are restricted to low-frequency events, which have decreased markedly since the 1980s (Frutos et al., 2004).

Under these circumstances, future study reach management plans have to include the goal of ecological restoration. Basin, reach and site scale restoration should ensure a hydrogeochemical diversity of RWs in the middle Ebro floodplains. Alternative strategies would need to make more integrated use of natural resources, primarily soil and water, as suggested by Comín (1999). Such management strategies should ensure erosive floods (> 3000 m^3 s^{-1}) that would restore geomorphological dynamics, at least to a certain extent, and also a range of seasonal discharge fluctuations below bankfull (1600 m^3 s^{-1}, 1927-2003). This would maintain a diverse array of conditions over multiple spatial and temporal scales. In addition, the influence of agricultural run-off on alluvial aquifer height and water quality should be addressed since it has been shown to affect both the river and floodplain water bodies (FNE, 1993; Torrecilla et al., 2005).

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REFERENCES

Amoros, C. & G. Bornette. 2002. Connectivity and biocomplexity in waterbodies of riverine floodplains. Freshwater Biology 47: 761-776.

Amoros, C., A. L. Roux, J. L. Reygrobellet, J. P. Bravard & G. Pautou. 1987. A method for applied ecological studies of fluvial hydrosystems. Regulated Rivers-Research & Management 1: 17-36.

APHA. 1989. Standard Methods for the Examination of Water and Wastewater. APHA, Washington D.C., USA.

Arntzen, E. V., D. R. Geist & P. E. Dresel. 2006. Effects of fluctuating river flow on groundwater/surface water mixing in the hyporheic zone of a regulated, large cobble bed river. River Research and Applications 22: 937-946.

Batalla, R. J., C. M. Gomez & G. M. Kondolf. 2004. Reservoir-induced hydrological changes in the Ebro River basin (NE Spain). Journal of Hydrology 290: 117-136.

Begueria, S. 2005. Uncertainties in partial duration series modelling of extremes related to the choice of the threshold value. Journal of Hydrology 303: 215-230.

Bornette, G., C. Amoros & N. L. Lamouroux. 1998. Aquatic plant diversity in riverine wetlands: The role of connectivity. Freshwater Biology 39: 267-283.

Brunke, M., E. Hoehn & T. Gonser. 2003. Patchiness of river-groundwater interactions within two floodplain landscapes and diversity of aquatic invertebrate communities. Ecosystems 6: 707-722.

Cabezas, A., E. Gonzalez, B. Gallardo, M. García, M. Gonzalez & F. A. Comín. in press. Effects of hydrological connectivity on sediment and understory structure in the middle Ebro river (NE Spain): Implications for restoration and management. Aquatic Sciences.

Castro, P., J. Guerrero & M. A. Muñoz. 2001. Restoration planning of the riparian forest at the Galachos Natural Reserve (Zaragoza). Consejo de Proteccion de la Naturaleza de Aragon, Zaragoza (in Spanish).

Comin, F. A. 1999. Management of the Ebro River Basin: Past, present and future. Water Science and Technology 40: 161-168.

Domitrovic, Y. Z. 2003. Effect of fluctuations in water level on phytoplankton development in three lakes of the Parana river floodplain (Argentina). Hydrobiologia 510: 175-193.

FNE-UZ. 1993. Contamination of the alluvial aquifer in the Ebro river corridor. Fundación Nueva Empresa, Zaragoza (in Spanish).

Frutos, L. M., A. Ollero, and M. Sánchez Fabre. 2004. Characterization of the Ebro river and its catchment and its hydrological fluctuations. In A. Gil Oncina (ed.), Alteración de los regimenes fluviales peninsulares. Fundación Caja Murcia, Alicante: 223-280 (in Spanish).

Galat, D. L., L. H. Fredrickson, D. D. Humburg, K. J. Bataille, J. R. Bodie, J. Dohrenwend, G. T. Gelwicks, J. E. Havel, D. L. Helmers, J. B. Hooker, J. R. Jones, M. F. Knowlton, J. Kubisiak, J.

Mazourek, A. C. McColpin, R. B. Renken & R. D. Semlitsch. 1998. Flooding to restore connectivity of regulated, large-river wetlands - Natural and controlled flooding as complementary processes along the lower Missouri River. Bioscience 48: 721-733.

Geerling, G. W., A. M. J. Ragas, R. Leuven, J. H. van den Berg, M. Breedveld, D. Liefhebber, & A. J. M. Smits. 2006. Succession and rejuvenation in floodplains along the river Allier (France). Hydrobiologia 565: 71-86.

Hancock, P. J. 2002. Human impacts on the stream-groundwater exchange zone. Environmental Management 29: 763-781.

Heiler, G., T. Hein, F. Schiemer & G. Bornette. 1995. Hydrological connectivity and flood pulses as the central aspects for the integrity of a river-floodplain system. Regulated Rivers-Research & Management 11: 351-361.

Hein, T., C. Baranyi, G. Heiler, C. Holarek, P. Riedler & F. Schiemer. 1999. Hydrology as a major factor determining plankton development in two floodplain segments and the River Danube, Austria. Archiv Fur Hydrobiologie 3: 439-452.

Hein, T., C. Baranyi, W. Reckendorfer & F. Schiemer. 2004. The impact of surface water exchange on the nutrient and particle dynamics in side-arms along the River Danube, Austria. Science of the Total Environment 328: 207-218.

Hohensinner, S., H. Habersack, M. Jungwirth & G. Zauner. 2004. Reconstruction of the characteristics of a natural alluvial river-floodplain system and hydromorphological changes following human modifications: The Danube River (1812-1991). River Research and Applications 20: 25-41.

Ibanez, C., N. Prat & A. Canicio. 1996. Changes in the hydrology and sediment transport produced by large dams on the lower Ebro river and its estuary. Regulated Rivers-Research & Management 12: 51-62.

Junk, W. J., B. Bayley & R. E. Sparks. 1989. The flood pulse concept in river-floodplain systems. In D. P. Dodge (ed.), Proc. Intern. Large River Symposium. Special Issue of Journal of Canadian Fisheries and Aquatic Sciences:11-27.

Knowlton, M. F. & J. R. Jones. 1997. Trophic status of Missouri River floodplain lakes in relation to basin type and connectivity. Wetlands 17: 468-475.

Lamers, L. P. M., R. Loeb, A. M. Antheunisse, M. Miletto, E. C. H. E. T. Lucassen, A. W. Boxman, A. J. P. Smolders & J. G. M. Roelofs. 2006. Biogeochemical constraints on the ecological rehabilitation of wetland vegetation in river floodplains. Hydrobiologia 565: 165-186.

Lamontagne, S., F. W. Leaney & A. L. Herczeg. 2005. Patterns in groundwater nitrogen concentration in the riparian zone of a large semi-arid river (River Murray, Australia). River Research and Applications 22: 39-54.

Losada, J. A., S. Montesinos, M. Omedas, M. A. Garcia & R. Galvan. 2004. Cartography of the February-2003 flood at the Ebro river: Performance of photo interpretation, remote sensing and SIG anlysis in the GIS-EBRO. In C. Conesa, Y. Alvarez & J. B. Martinez (eds.), Metods Cuantitativos, SIG y Teledeteccion. Universidad e Murcia, Murcia. : 207-218 (in Spanish).

Macpherson, G. L. & M. Sophocleous. 2004. Fast ground-water mixing and basal recharge in an unconfined, alluvial aquifer, Konza LTER Site, Northeastern Kansas. Journal of Hydrology 286: 271-299.

Malard, F., K. Tockner & J. V. Ward. 2000. Physico-chemical heterogeneity in a glacial riverscape. Landscape Ecology 15: 679-695.

Malard, F., U. Uehlinger, R. Zah & K. Tockner. 2006. Flood-pulse and riverscape dynamics in a braided glacial river. Ecology 87: 704-716.

McClain, M. E., E. W. Boyer, C. L. Dent, S. E. Gergel, N. B. Grimm, P. M. Groffman, S. C. Hart, J. W. Harvey, C. A. Johnston, E. Mayorga, W. H. McDowell & G. Pinay. 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6: 301-312.

Metzger, J. P. & H. Décamps. 1997. The structural connectivity treshold : an hypothesis in conservation biology at landscape scale. Acta Oecologica 18: 1-12.

Ollero, A. 1992. The free meanders of the middle Ebro (Logroño-La Zaida): Fluvial geomorphology, eco-geography and risk analysis. PhD dissertation. University of Zaragoza, Zaragoza, Spain (in Spanish).

Ollero, A. 1995. Dynamic of the Ebro main channel at the Galachos natural reserve. Cuaternario y Geomorfología 9: 85-93 (in Spanish).

Ollero, A. 2007. Channel adjustments, floodplain changes and riparian ecosystems of the middle Ebro River: Assessment and management. International Journal of Water Resources Development 23: 73-90.

Pinilla, V. 2006. The development of irrigated agriculture in twentieth-century Spain: a case study of the Ebro basin. Agricultural History Review 54: 122-141.

Regato, P. 1988. Contribution to the study of flora and vegetation of La Alfranca oxbow lake and its relationship with the evolution of the fluvial system, Diputación General de Aragon, Zaragoza. (in Spanish).

Sánchez-Pérez, J. M., P. Vervier, F. Garabétian, S. Sauvage, M. Loubet, J. L. Rols, T. Bariac & P. Weng. 2003. Nitrogen dynamics in the shallow groundwater of a riparian wetland zone of the Garonne, SW France: nitrate inputs, bacterial densities, organic matter supply and denitrification measurements. Hydrology and Earth System Sciences 7: 97-117.

Tabacchi, E., D. L. Correll, R. Hauer, G. Pinay, A. M. Planty-Tabacchi & R. C. Wissmar. 1998. Development, maintenance and role of riparian vegetation in the river landscape. Freshwater Biology 40: 497-516.

Thomaz, S. M., L. M. Bini & R. L. Bozelli. 2007. Floods increase similarity among aquatic habitats in river-floodplain systems. Hydrobiologia 579: 1-13.

Tockner, K., D. Pennetzdorfer, N. Reiner, F. Schiemer & J. V. Ward. 1999. Hydrological connectivity, and the exchange of organic matter and nutrients in a dynamic river-floodplain system (Danube, Austria). Freshwater Biology 41: 521-535.

Tockner, K. & J. A. Stanford. 2002. Riverine flood plains: present state and future trends. Environmental Conservation 29: 308-330.

Torrecilla, N. J. 2002. Hydrogeology and Hydroenvironment of the Alfranca oxbow lake and surroundings. Departamiento Medio Ambiente, DGA, Zaragoza, Spain (in Spanish).

Torrecilla, N. J., J. P. Galve, L. G. Zaera, J. F. Retarnar & A. N. A. Alvarez. 2005. Nutrient sources and dynamics in a mediterranean fluvial regime (Ebro river, NE Spain) and their implications for water management. Journal of Hydrology 304: 166-182.

Tremolieres, M., I. Eglin, U. Roeck & R. Carbiener. 1993. The Exchange Process between River and Groundwater on the Central Alsace Floodplain (Eastern France) .1. The Case of the Canalised River Rhine. Hydrobiologia 254: 133-148.

Valett, H. M., M. A. Baker, J. A. Morrice, C. S. Crawford, M. C. Molles, C. N. Dahm, D. L. Moyer, J. R. Thibault & L. M. Ellis. 2005. Biogeochemical and metabolic responses to the flood pulse in a semiarid floodplain. Ecology 86: 220-234.

Vandenbrink, F. W. B., J. P. H. M. Deleeuw, G. Vandervelde & G. M. Verheggen. 1993. Impact of Hydrology on the Chemistry and Phytoplankton Development in Floodplain Lakes Along the Lower Rhine and Meuse. Biogeochemistry 19: 103-128.

Ward, J. V. 1989. The 4-Dimensional Nature of Lotic Ecosystems. Journal of the North American Benthological Society 8: 2-8.

Ward, J. V. 1998. Riverine landscapes: Biodiversity patterns, disturbance regimes, and aquatic conservation. Biological Conservation 83: 269-278.

Ward, J. V. & J. A. Stanford. 1995. Ecological Connectivity in Alluvial River Ecosystems and Its Disruption by Flow Regulation. Regulated Rivers-Research & Management 11: 105-119

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STUDY III

Effects of hydrological connectivity on the substrate and understory structure of riparian wetlands in the Middle Ebro River (NE Spain): Implications for restoration and management

Effects of hydrological connectivity on the substrate and understory structure of riparian wetlands in the Middle Ebro River (NE Spain): Implications for restoration and management

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Abstract

The hydroperiod, flooded sediments and riparian understory were examined for eight riparian wetlands of one Ebro River reach (NE Spain) to relate river-floodplain interactions at different spatio-temporal scales with wetland strucuture. This analysis served as a basis for assessing the ecological status of the study reach and proposing a valid restoration plan. A comparison of water-level fluctuations in riparian wetlands with that in the river channel during an ordinary flood was use to characterize the hydroperiod. This characterization was further linked with the results of a multivariate analysis performed using sediment physico-chemistry. Moreover, different measures of understory diversity were used to estimate the successional stage of eight riparian wetlands located in the same reach of the Ebro River. We described four hydroperiod types from the examined flood, from disconnected oxbow lakes to backflow channels fluctuating in concert with the Ebro River. Also three types of sediment differ mainly differing in their organic matter content. Both results were closely related reflecting the dominance of endogenous or allogeneous processes. However, such heterogeneity was interpreted as being variable over longer spatio-temporal scales. In addition, the riverscape was found to be homogenous and dominated by wetlands at mature successional stages. Consequently, the lack of erosive floods within the reach seems to make inclusion of ecological restoration of geomorphological dynamics a highly appropriate management objective. Alternative strategies at both reach and site scales are proposed.

Keywords: riverscape, hydroperiod; hysteretic loops; sediment; understory; Ebro.

Introduction

River floodplains are flood-dependent ecosystems that are an integral part of the river (Ward, 1998). Natural disturbances caused by floods create and maintain a complex mosaic of riparian landforms and associated aquatic and semi-aquatic communities, while hydrological connectivity promotes the exchange of matter and energy between different parts of the river system (Junk et al., 1989; Ward, 1989). Hydrogeomorphic variables therefore establish the physical template and provide constraints under which chemical and biological processes operate (Tabacchi et al., 1998). Focussing on the riverscape (*sensu* Malard et al., 2000), the interactions of processes at different spatio-temporal scales promote a combination of complex gradients of habitat conditions, which can result in high levels of diversity (Amoros and Bornette, 2002).

Anthropogenic alterations of river flows and floodplains often disrupt the intensity, frequency, and timing of the natural disturbance regime responsible for maintaining the ecological integrity of these ecosystems (Ward and Stanford, 1995). In the developing world, the remaining natural floodplains are disappearing at an accelerating rate, primarily because of changing hydrology (Tockner and Stanford, 2002). Consequently, the conservation of these ecosystems depends on the rehabilitation of river-floodplain interactions at several spatio-temporal scales (Henry and Amoros, 1995; Tockner et al., 1998; Hughes et al, 2001; Brunke, 2002).

On the inter-annual scale, erosive floods create and maintain a diversity of successional stages that determine the overall complexity of the landscape matrix (Metzger and Décamps, 1997; Galat et al., 1998). In the absence of natural disturbances, the floodplain system probably tends toward geographical and temporal uniformity (Tockner et al., 1998). Under these conditions, the riverscape becomes dominated by mature stages since successional pathways can proceed unimpeded and new wetlands are not being created (Ward and Stanford, 1995). Within water bodies, the presence of a wide array of environmental conditions fosters a high level of biodiversity, although it is necessary to include measures such as the species turnover rate for a comprehensive understanding of natural patterns and processes (Ward et al., 2001). In the present study, different estimates of understory diversity were employed to infer wetland successional stage. This analysis, which relies on the close relationship of riparian vegetation with wetland topography, serves as a basis for assessing the ecological status of the riverscape at the reach scale.

On a seasonal time scale, variations in hydrological connectivity are caused by water level fluctuations (Tockner et al., 2000). The wetland hydroperiod concept, introduced by Mitsch and Gosselink (1993), has been used to compare hydrological connectivity between wetlands in relation to water level fluctuations. In our study, sediment structure was used to relate hydroperiod measurements with the balance between autogeneous or allogeneous processes on sediment diagenesis. Rostan et al. (1987) emphasized the importance of sediment characteristics in determining the relative dominance of endogenic and exogenic processes. In addition, Tockner and Schiemer (1997) use the organic matter of wetlands substrate to indicate a decrease in natural disturbances.

Previous studies have shown that the geomorphology and vegetation of the study reach has been strongly modified by alteration of the fluvial dynamic (Regato, 1988; Castro et al., 2001; Ollero, 2007). Successful application of the Water Framework Directive critically depends on floodplain restoration. To achieve this goal will require implementation of adequate ecological restoration projects, which are more likely to be successful if based on an understanding of the underlying geomorphological and ecological processes (Kondolf, 1998). The objectives of this study are: a) to relate sediment and riparian understory structure with hydrological connectivity in a variety of riparian wetlands within a reach of the Middle Ebro River, b) to assess the ecological status of the study reach, and c) to characterize the implications of these findings with respect to the ecological restoration and management of the study reach.

STUDY AREA

Study reach

The study area (Fig. 1) is in the Middle Ebro River in northeast Spain and comprises a watershed area of 85,362 km^2. The Ebro River, 910 km long, is the largest river in Spain. It has an annual discharge into the Mediterranean Sea of 18,138 hm^3/y and remains geomorphologically active despite the presence of 170 dams and reservoirs on the river and its tributaries. The section of the Ebro River in the study area is a meandering reach (sinuosity = 139, slope of the bankfull channel = 0.05%, average floodplain width = 5 km). At the study reach, the discharge, averaged over the years 1927 to 2003, is 230 m^3/s and elevation ranges from 175 m asl in the river channel to 185 m asl at the base of the old river terrace. The area flooded by the 10-yr return period flood (3000 m^3/s, 1927-2003) is 2230 ha, although only about 14% of the area is flooded by a river discharge of 1000 m^3/s (0.37 y return period, 1927-2003), and only 4% is flooded by a river discharge of 600 m^3/s (0.14 y return period, 1927-2003). During the last century, the number and extent of permanent water bodies has declined considerably.

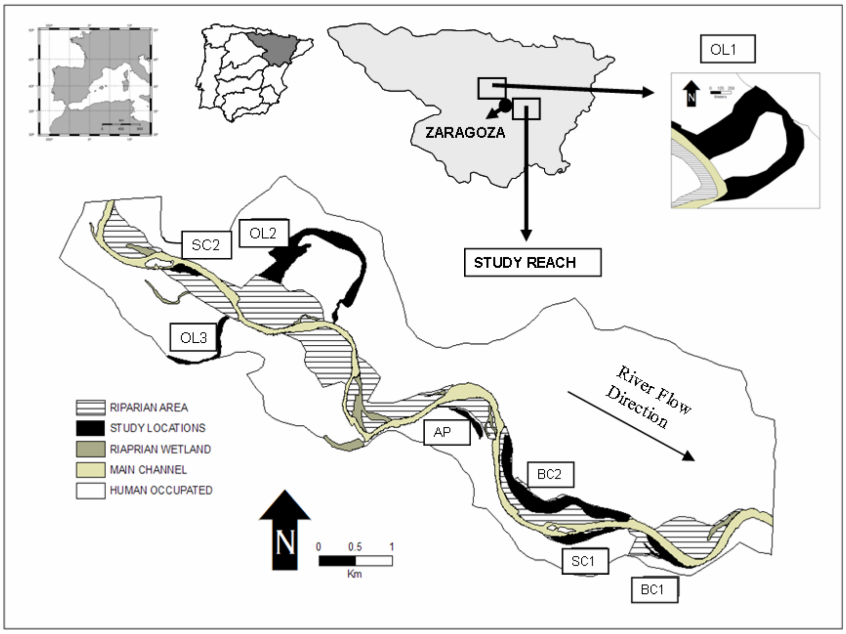

Figure 1. Location of the selected Riparian Wetlands at the study reach delimited by the 10-year floodplain. Only one of the wetlands (OL1 in Fig. 2), located approximately 3 km upstream of Zaragoza city, was outside of this area.

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Riparian wetlands

Most of the riparian wetlands examined are within the study reach, 12 km downstream from the city of Zaragoza. An exception was Juslibol oxbow lake (OL1 in Fig. 1), which is 1 km upstream of the city. At the study reach, only 3.6% of the area flooded by a 10-yr return period flood is occupied by riparian wetlands (Fig. 1). Most of the remaining area is used for agriculture. The study sites cover 70% of the riparian wetland area, so they are assumed to faithfully reflect the state of the riverscape in the area. These sites represent the widest range of hydrological connectivity (in type and magnitude) in the study reach. Connectivity types can vary depending on river discharge, as is the case with the BC2 site (Fig. 2). BC2 is a large side-channel during high magnitude flows; however, during ordinary floods its upstream and downstream areas function as side and backflow channels, respectively. The age of each wetland since creation (Table 1) was estimated from time of first observance based on aerial photography, using 1927, 1957, 1981, 1998 and 2003 ortophotos.

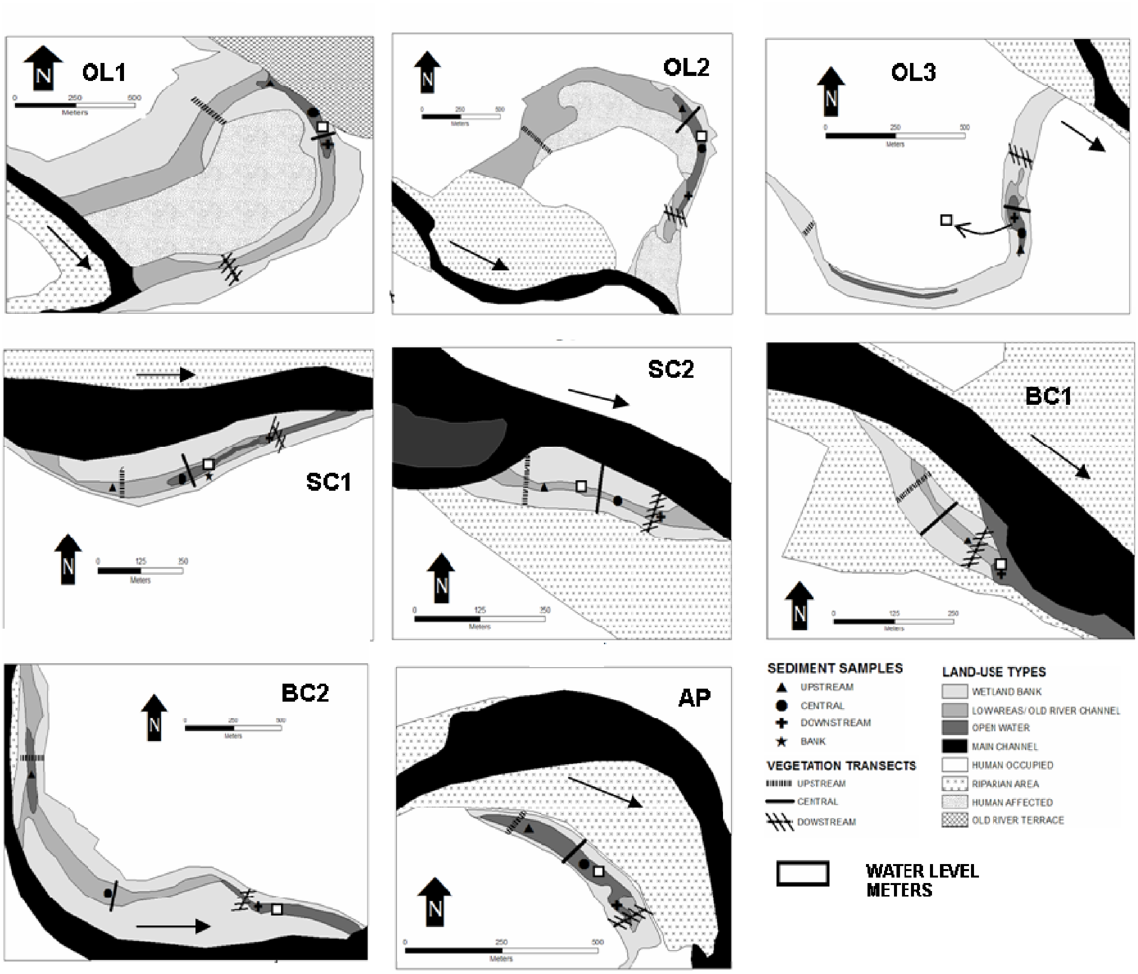

Figure 2. Geomorphological features, location of transects where plant studies were performed, sediment sampling points and location of water level meters for each study site. Arrows in each box indicate the direction of Ebro river flow. Land use cover types are shown. "Human Affected" includes old gravel pits and abandoned agricultural fields, while "Human Occupied" represents present agricultural fields, paths and urban settlements. Transect length is shown for each wetland in figure 6 (at right).

DESCRIPTION			HYDROPERIOD		AREA (Ha)			
Label	**Type**	**Age (y)**	**SWL(m)**	**WLV(cm)**	**RB**	**OW**	**ORC**	**Total**
AP1	Artificial Pond	11	1.34	76.5	2.64	1.89	0.30	4.82
OL1	Oxbow lake	42	0.8	65.5	48.66	2.31	19.34	70.31
OL2	Oxbow lake	61	1.73	20.3	5.91	3.90	25.65	35.45
OL3	Oxbow lake	61	2.42	19.9	8.92	1.06	0.35	10.33
SC1	Side channel	34	0.7	119.1	0.41	0.10	0.19	0.69
SC2	Side channel	34	0.1	142.3	2.53	0.00	0.67	3.20
BC1	Backflow channel	61	0.33	118.5	2.18	1.39	0.55	4.13
BC2	Backflow channel(1)	34(1)	0.65	130	42.14	4.41	10.11	56.66

(1)During low-magnitude events, BC2 is a backflow channel at its downstream part (34 years) but can be defined as a younger (11 years) side channel at its upstream end.

Table 1. Characteristics of the studied riparian wetlands by type, hydroperiod, age and area. SWL, Water Level in the deepest part in the summer sampling; WLV, Water Level fluctuation during the studied ordinary flood; RB, Riparian Bank; OW, Open Water; ORC, Old River Channel.

Methods

Hydroperiod measurements

Water levels in the riparian wetlands were measured hourly during an ordinary flood that peaked on April 24, 2006, reaching a flow rate of 536 m^3/s (0.15 y return period, 1927-2003) at the Zaragoza gauging station 12 km upstream of the study area. To measure water levels at Alfranca Oxbow (OL2 in Fig. 2), we used a shaft encoder with 1-mm resolution (Thalimedes, OTT-Hydrometry®) and an integrated data logger. In the other riparian wetlands, pressure-based meters with 1-cm resolution (DI241 Diver, Van Essen Instruments®) were used. The monitoring network of the Ebro River Basin Administration (www.chebro.es) provided hourly data recorded at the Zaragoza gauging station.

To compare the magnitude of changes in water level among all of the riparian wetlands, the readings between 22/4/06 and 30/4/2006 were converted to relative water level (RWL) values using the height reading and the full range of water level variation (WLV) at each site during the examined flood (Table 1). To compare their dynamics during the flood, the RWLs at each site and in the Ebro River were plotted together (Fig. 3, left column). A plot of the RWL of each site versus the RWL of the river (Fig. 3, right column) revealed hysteretic loops, demonstrating the degree of connectivity in study site and river RWL dynamics.

Sediment Collection and Analysis.

We chose to measure sediment physico-chemical variables and relate them to the wetland hydroperiod because they can provide valid indicators of hydrological connectivity type and magnitude (Tockner and Schiemer, 1997; Bornette et al., 2001). Sampling was performed at flooded areas because these areas are affected by the widest range of flood magnitudes, and because this sampling strategy allows for inter-wetland comparisons. Moreover, two different seasons with contrasting river discharges were selected for sampling because of the possible effect of flooding on physical, chemical and biological processes that affect sediment variables. Sediment samples were collected in February 2005 (n = 22 samples) after two recurrent floods of 700 m^3/s, and in August (n = 21 samples) after two months of low water levels, at the upstream, central, and downstream section of the flooded zone of the eight examined wetlands (Fig. 2, Upstream [U[, Central [C], Downstream [D], and Bank [B]). Top-sediment cores (0-3 cm) were collected directly into transparent PVC tubes (∅ = 46 mm) or, where the water depth prevented direct collection, using an Ekman grab sampler. The samples were transported to the laboratory in dark cool-boxes and processed immediately.

Sediment pH and conductivity were measured in a solution of 10 g of fresh sediment dispersed in deionized water (pH: 2.5:1 g/ml, conductivity: 5:1 g/ml) after shaking for 30 min. A 4-g subsample of fresh sediment was mixed with 40 ml of KCl 0.01 N in a 50-ml wide-mouth flask, gently shaken for 30 min, and centrifuged at 2500 rpm. The supernatant was filtered through filter paper (Whatman© no. 42) and the solution was stored at -20ºC. Within two weeks of collection, the samples were analyzed for nitrate (NO_3^-), ammonium (NH_4^+), phosphate (PO_4^{3-}), sulphate ($SO_4^=$), calcium (Ca^{2+}), magnesium (Mg^{2+}), and sodium (Na^+) using a high-performance liquid chromatography (HPLC) analyzer. To mimic the mean conductivity values of river water at the study sites and to avoid interfering with ion chromatography analysis, extractions were performed using a dilute salt solution (KCl 0.01 N; 1413 μS). Ion concentrations, expressed in ppm (mg/kg fresh sediment), were thus representative both of pore-water and the easily extractable fraction in the top-sediment layer. To determine dry mass and percentage moisture (by mass) on a wet-weight basis, the remaining fresh sediment was oven-dried at 60ºC. Bulk density was calculated as dry mass by volume (g/cm^3), and organic matter was estimated by loss on ignition (LOI) in a muffle furnace (450ºC for 5.5 h), and expressed as percentage of sediment dry mass.

Differences between high and low water levels were determined by individually subjecting winter and summer data to multivariate statistical analysis that incorporated Principal Components Analysis (PCA) with the Factor analysis procedure, and Cluster Analysis based on a Ward algorithm (SPSS© 14.0 package). PCA was used to identify factors that accounted for data variability as well as to categorize sites according to those factors. The initial PCA solution was varimax-rotated, and the most suitable number of factors, in terms of explanatory power and interpretation, was extracted. The resulting scores were recorded for each sampling point. Sediment types were determined by testing and interpreting PCA results using winter and summer cluster solutions. A one-way ANOVA was used to detect differences in variables included in the multivariate analysis between sediment types. Phosphate was not included in the statistical analyses because in a majority of the samples the concentration of phosphate in the extracted solution was less than the HPLC detection threshold (0.005 mg/l). Where variables were not normally distributed, the data were transformed before inclusion in analysis.

Understory vegetation sampling and analysis

In August 2005, three transects were placed at the upstream, central, and downstream sections of the riparian wetland sites (Fig. 2). Transect length (55-300 m) was delimitated by human landscape features (e.g., paths and crops) for the oxbow lakes and the artificial pond (OL2, OL3, OL1 and AP). For the outer banks of side channels (SC1, BC1, BC2 and SC2), transect length was delimitated by abrupt topographic changes, whereas the limit at sides adjacent to the riverbank was marked by point bars associated with the main channel. At side channels, these limits nearly coincided with the area flooded by a 1500 m^3/s event (0.87 y flood, 1927-2003).

Sampling plots were set every 5 m along transects (n = 11-50). For each plot, the ground cover for each understory species was visually estimated within a 1-m^2 quadrant. Species were identified according to Aizpuru et al. (2003). Cover values were normalized to a 0-1 range to take into account all considered vegetation layers per plot. In previous years, submerged macrophytes were uncommon in the riparian wetlands therefore ground cover was considered null in the open-water plots. The visual cover in flooded plots, mostly occupied by the emergent macrophytes *Phragmites australis* and *Typha* sp., was estimated. Common understory types and their representative species are shown in Table 2.

Cumulative Diversity (CD) spectra were drawn to explore the effects of geomorphological features on the riparian understory structure. Both wetland banks were thought to provide useful information, so transects were

Site description	Representative species
Wetland banks frequently flooded	*Paspalum paspalodes* (Michaux) Scribner *Xanthium equinatum* Murray subsp. *italicum* (Moretti) O. Bolòs & J.Vigo
Flooded plots Muddy areas with high water tables	*Phragmites australis* (Cav) Trin. Ex Steudel *Typha* L.
Elevated areas close to open water plots Accreted areas ; Intermediate flooding	*Atriplex prostata* Boucher ex DC *Galium Palustre* L. *Phalaris arundinacea* L. *Rubia tinctorium* L. *Rumex conglomeratus* Murray *Rumex Crispus* L. *Torilis arvensis* subs. *arvensis* (Hudson) Link
Low areas ; Intermediate flooding Depressed areas far from open water plots	*Braquipodium sylvaticum* subs. *sylvaticum* (Hudson) Beauv. *Carex divisa* Hudson *Elymus repens* (L.) Gould subsp. *repens* *Hedera helix* L. *Juncus acutus* L.
Mature riparian forest; Rarely flooded	*Arundo donax* L. *Cynanchum acutum* L. *Rubus ulmifolius* Schott *Solanum dulcamara* L.
Accreted areas far from open water plots Rarely flooded	*Atriplex patula* L. *Avena Barbata* Pott ex Link subsp. *barbata* *Foenicolum vulgare* Miller subsp. *vulgare* *Glycyrrhiza glabra* L.

Table 2. Representative understory species found at the most common plot types in relation to their hydrological connectivity. Species identification and nomenclature are according to Aizpuru et al. (2003).

divided into two different sections (left and right), positioning the starting point at the deepest part of each wetland. The cumulative ground-cover values determined successively, from the starting plot, were normalized to 0-1 and used to calculate the understory cumulative diversity (CD) for each plot using the Shannon Index (H); both left and right sections were plotted (Fig. 6, left column).

Additional characterizations of wetland successional stage were provided by estimating diversity and environmental heterogeneity for each section, using the riparian understory. Since it integrates ground-cover values of all section plots, as described above, the CD of the most outer plot was taken as the measure of each section's diversity. Environmental heterogeneity for each section was evaluated by estimating a Cumulative Diversity Rate (CDR) using CD values (diversity units) and section length (meters) as follows:

$$CDR = (CD_{final} - CD_{starting\ point})/\text{Section Length}$$

CD was then plotted as a function of CDR for each section (right column in Fig. 6). The information provided by CD and CDR was considered similar to gamma and beta diversity at the section scale. Low species turnover rates lead to homogenization and low CDR values, while low CD values result if environmental conditions are such that they prevent species from establishing along the section.

Results

Types of hydroperiod

The study sites differed in their types of connectivity as reflected in the variation in RWLs and hysteretic loops (Fig. 3). At BC1 and BC2, water levels rose and fell in concert with the river water, and their hysteretic loops had diagonal forms. At OL1 and SC1, however, changes in RWL were delayed relative to those of the Ebro River (Fig. 3), which is reflected by an inflexion point in their hysteretic loops during the rising phase. However, they differed during their receding phases. Note that the upstream section of BC2 is a connected side channel for which the hysteretic loop is probably more similar to that of the SC1 site than is suggested by its downstream end where the pressure-based meter was located. The hysteretic loops for SC2 and AP were curved. At SC2, the peak in RWL coincided with the timing of maximum river discharge, but at AP, it was delayed relative to that of the river. At AP, the hysteretic loop was round during both the rising and receding flood phases, whereas the hysteretic loop for SC2 showed this round form only when river discharge decreased. For OL2 and OL3, the RWLs were not correlated with that of the river.

Sediments

Physico-chemical characteristics of riparian sediments

The major components of the PCA explained 68% and 64% of the overall variance in winter and summer, respectively (Table 3). The variation in sediment characteristics of the sampling sites was most strongly correlated with bulk density, organic matter and ammonia content. This was valid for both analyzed seasons, although pH, conductivity, sulphate, and sodium also had high factorial loadings on the first factor in winter. The second and third principal components explained between 16% and 20% of the total variance. In winter, the second principal component included concentrations of calcium, magnesium, and nitrates. In summer, nitrate and sulphate concentrations were associated with the second axis, and calcium and magnesium concentrations were associated with the third axis.

In winter, the first factor clearly discriminated the locations of the disconnected oxbow lakes (OL3 and OL2 in Fig. 2), and also that of the backflow area of two former river channels (BC2-U and BC1-D in Fig. 2), which had more organic, salty and ammonia-enriched substrates than did all other locations (Fig. 4). Moreover, OL3 locations presented extraordinarily high values for factor 2, as did two locations of a connected oxbow lake (OL1 in Fig. 2) and the central part of an intermediate connected side channel (SC2-C in Fig. 2). In summer, differences in factor 1 persisted, although being slightly attenuated. A suite of substrates found dried in summer presented elevated values for factor 2, conjointly with those for sediments from OL3 locations (Fig. 5).

Sediment types

Three types of riparian sediments were identified on the basis of PCA results and separated according to location groups by Cluster analysis (Fig. 5): a) *Organic Sediments (ORGSED, n = 6):* Locations clustered as organic (ORG) during winter and summer. This type comprised locations of disconnected oxbow lakes (OL3 locations, OL2-U and OL2-C) and one accreted backflow channel (BC1-D). The sediments in this type presented the highest values for organic matter, conductivity and the

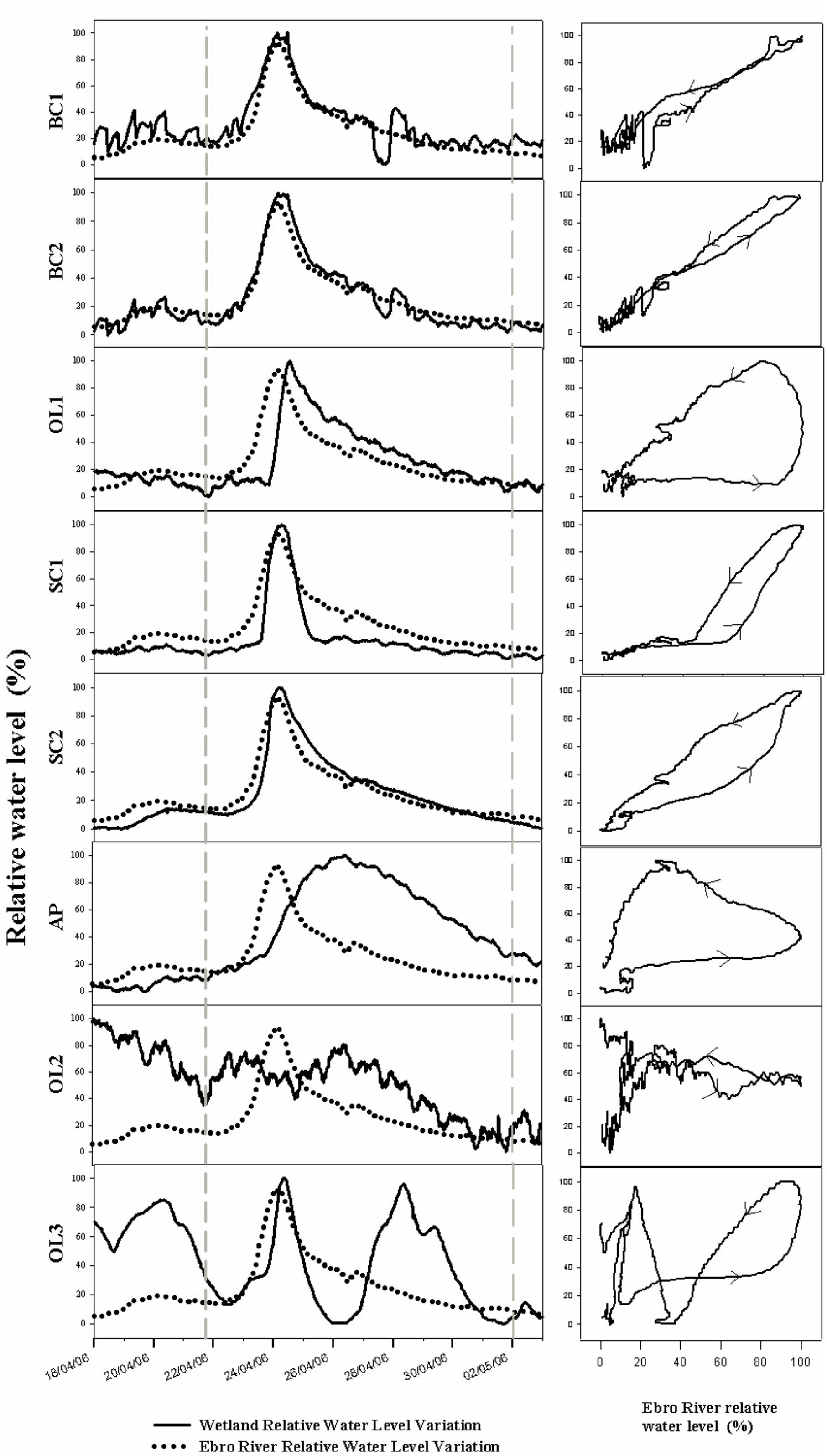

Figure 3. Comparison of RWL between the Ebro River and the studied wetlands during an ordinary flood (536 m3/s). RWL was calculated using the real water height and the full range of water level variation (WLV in Table 1) for each wetland. Dashed lines delimitate the period under consideration for the Hysteretic loops, which are displayed on the right part of the figure.

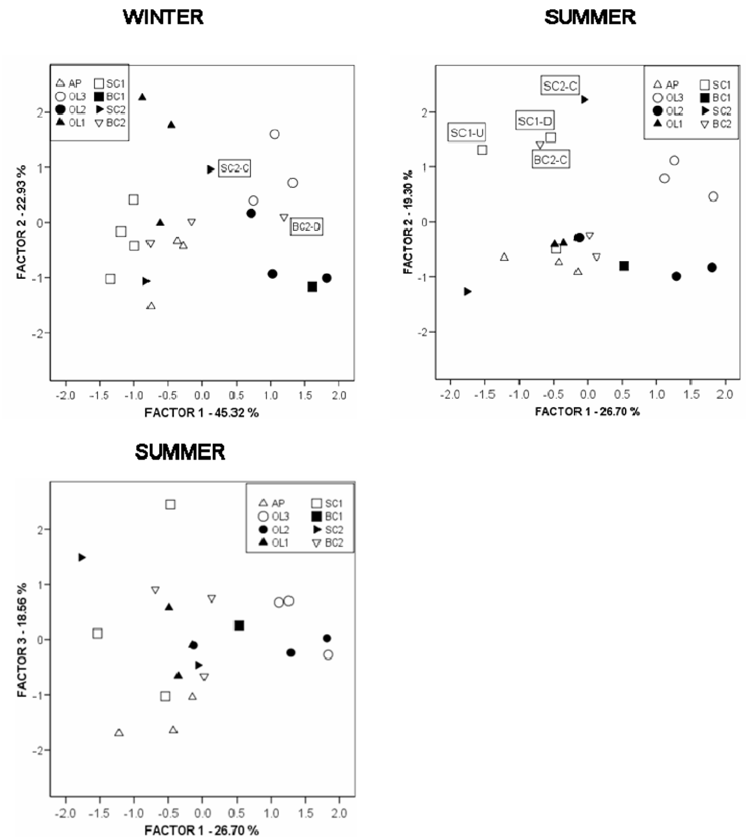

Figure 4. Position of sediment sample points on PCA considered as factors in winter and summer. Factor loading of the variables related to the different factors are displayed in Table 5. Locations have been grouped with Cluster results: OL3, Carttuja; SDS, Summer Disconnected Sediments; DS, Disconnected Sediments; CIS, Connected Inorganic Sediments; COS, Connected Organic Sediments. b) Highlighted locations are depicted in figure 2.

Season	Factor	Accumulated Variance	Explained Variance	Related Variables	Factor Loading
Winter	1	45.32	45.32	Na^+	0.92
				Conductivity	0.85
				Organic Matter	0.81
				Bulk Density	-0.80
				$SO_4^=$	0.67
				pH	-0.66
				NH_4^+	0.65
	2	68.26	22.93	Ca^{++}	0.87
				NO_3^-	-0.68
				Mg^{++}	0.67
Summer	1	26.70	26.70	Bulk Density	-0.84
				Organic Matter	0.82
				NH_4^+	0.79
	2	45.99	19.30	$SO_4^=$	0.94
				Conductivity	0.71
				NO_3^-	0.60
	3	64.65	18.66	Mg^{++}	0.94
				Ca^{++}	0.90

Table 3. Factor loading of sediment variables for the main factors of the PCA analysis for each period. Only high factor loadings have been included in the table for ease of interpretation.

lowest bulk densities for the two study periods. Consequently, their first factor scores in both PCA analyses were also the highest (Fig. 4). Inorganic nitrogen was present mainly as ammonia, which was found at the highest concentration in this location type in both summer and winter (Tables 4,5). b) *Intermediate Sediments (INTSED, n = 10):* Locations clustered as mineral (MIN) in winter and MIN-2 in summer, and locations clustered as ORG in winter and MIN in summer. This type comprised locations of the artificial pond (AP), a connected oxbow lake (OL1), as well as locations at two former side channels (SC2-C, BC2-U and BC2-D) and the downstream part of a disconnected oxbow lake (OL2-D). The sediments in this type presented intermediate values for conductivity and bulk density for the two study seasons (Tables 4,5). Their scores in the first factor of the PCA were intermediate in winter, as was the organic matter content, but lower in summer during which the percentage of organic matter was also markedly lower than in the ORGSED group (Fig. 4). Inorganic nitrogen was present mainly as ammonia, which was found at lowest concentrations in this location type in summer and at a concentration intermediate between the other two types in winter. The proportion of ammonia relative to total nitrate was much lower than for ORGSED (Tables 4,5). c) *Mineral Sediments (MINSED, n = 6):* Locations clustered as MIN in winter and either MIN-1 or MIN-3 in summer. This type comprised locations at the connected side channel (SC1) as well as locations at two former side channels (SC2-D and BC2-C). The sediments in this type presented the lowest conductivity values and the highest bulk densities for the two study periods. Nitrate and ammonia were in balance in winter, whereas ammonia prevailed in summer, although its proportion relative to total nitrate was much lower than for ORGSED (Tables 4,5). The first factor scores of both PCA analyses for this type were the lowest in winter, as was the organic matter content, but intermediate in summer, when the OM was as high as the INTSED group in summer.

Understory Structure

The morphology of the riparian wetlands significantly influenced the Cumulative Diversity (CD) Spectra (Fig. 6, left section). The spectra of the OL2 and OL1 sites exhibited similar patterns. Extended planar sections were observed in their central (CEN) and upstream (UP) spectra. In addition, bank communities displayed asymptotic shapes, although the right sections of the UP and DOWN spectra tended to be more linear. The AP and OL3 sites were characterized by rectangular-shaped spectra at the CEN and UP transects, although the spectra were more linear at their DOWN sites. Among the side channels, SC2 exhibited UP and DOWN spectra with diagonal forms, as did the UP spectra of SC1. At the DOWN and CEN sections of SC1 and at the CEN section of SC2, the spectral curves were asymptotic with uniform central areas. At the accreted back-flow water channel (BC1 in Fig. 2), the spectral curves were asymptotic. Only the DOWN spectra was *Phragmites*-dominated at the central section, which resulted in a vegetation structure that was less diverse than in the UP and CEN spectra. Similar patterns were present in the UP and DOWN spectra in BC2 (Fig. 2), although more homogeneous bank communities predominated. Intensive sheep grazing substantially altered the natural vegetation patterns in the central transect.

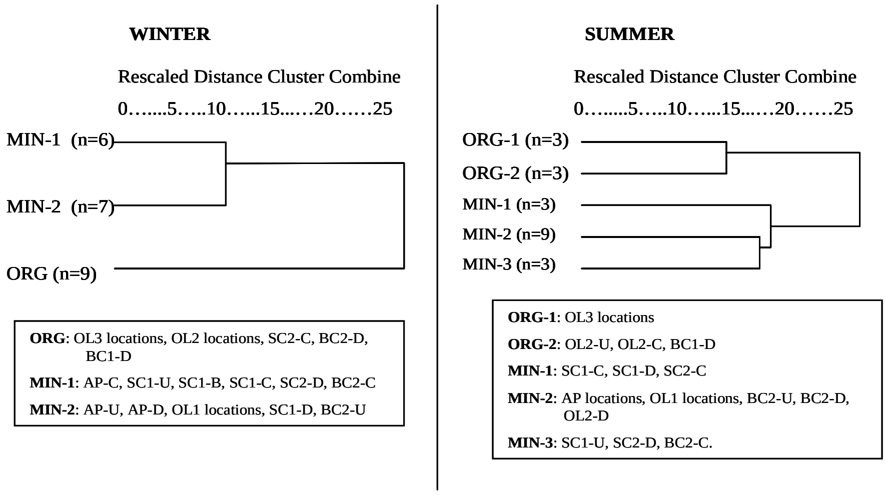

Figure 5. Locations dendrogram from winter and summer Cluster analysis. The sampled sites have been grouped for ease of interpretation: MIN, Mineral; ORG, Organic; U, Upstream; C, Central; D, Downstream; B, Bank

	ORGSED	INTSED	MINSED	P-VALUE
pH	7.88±0.11	7.88±0.14	8.57±0.10	0.003**
Cond (*u*s/cm^2)	409.50±71.74	239.27±42.69	80.42±9.67	0.001**
BD (g/cm^3)	0.25±0.04	0.54±0.07	1.02±0.06	0.000**
OM (%)	8.72±1.12	4.11±0.39	2.61±0.48	0.000**
NO_3^- (ppm)	1.10±0.66	0.70±0.35	3.01±1.23	0.385
$SO_4^=$ (ppm)	517.21±127.62	131.72±24.06	89.26±24.15	0.003**
Na^+ (ppm)	204.15±23.75	97.58±19.91	52.56±5.07	0.000**
NH_4^+ (ppm)	22.36±4.89	11.58±2.43	4.57±1.74	0.006**
Ca^{++} (ppm)	419.22±53.98	407.21±41.18	347.67±16.92	0.507
Mg^{++} (ppm)	61.40±5.19	53.17±7.39	37.90±2.19	0.087
FACTOR 1	1.26±0.16	-0.15±0.21	-1.02±0.09	0.000**
FACTOR 2	-0.06±0.46	0.29±0.35	-0.44±0.23	0.375

Table 4. Winter locations: mean values and standard error for sediment variables included in the multivariate analysis. Locations have been grouped according to Cluster results. Significance values for ANOVA are shown. OM, Organic Matter; BD, Bulk Density; Cond, Conductivity; ORGSED, Organic Sediments; INTSED, Intermediate Sediments; INORGSED, Inorganic Sediments. *$p < 0.05$; **$p < 0.01$.

	ORGSED	INTSED	MINSED	P-VALUE
pH	7.91±0.14	8.09±0.09	8.05±0.10	0.511
Cond (*u*s/cm^2)	354.17±29.82	301.70±19.40	282.20±31.54	0.192
BD (g/cm^3)	0.23±0.05	0.53±0.05	1.03±0.17	0.000**
OM (%)	10.69±1.49	4.10±0.63	4.07±0.46	0.000**
NO_3^- (ppm)	0.22±0.08	0.28±0.16	1.33±0.79	0.100
$SO_4^=$ (ppm)	487.56±190.74	344.15±139.43	740.58±245.95	0.334
Na^+ (ppm)	234.55±20.41	181.53±19.60	219.98±107.37	0.686
NH_4^+ (ppm)	23.61±3.96	4.43±1.55	5.97±2.99	0.000**
Ca^{++} (ppm)	416.84±57.08	371.64±34.91	635.97±92.38	0.012*
Mg^{++} (ppm)	70.64±3.06	57.42±5.37	72.36±9.78	0.172
FACTOR 1	1.30±0.19	-0.03±0.38	0.18±0.17	0.000**
FACTOR 2	-0.28±0.12	-0.23±0.28	-0.50±0.26	0.428
FACTOR 3	-1.00±0.27	0.50±0.57	0.78±0.59	0.44

Table 5. Summer locations: mean values and standard error for sediment variables included in the multivariate analysis. Locations have been grouped according to Cluster results. Significance values from ANOVA are shown. OM, Organic Matter; BD, Bulk Density; ORGSED, Organic Sediments; INTSED, Intermediate Sediments; INORGSED, Inorganic Sediments. *$p < 0.05$; **$p < 0.01$.

With respect to biodiversity measures, values at the CDR and CD were generally below the mean values at the OL3, OL2, and BC2 sites (Fig. 6, right). Only the UP sections of BC2, and the right section of the CEN and UP transects in OL2 had higher values. On the left bank of AP and BC1 sections, the understory was characterized by slightly higher CDR values. With the exception of the right section of BC1 at the upstream area, CD values at BC1 and AP sites were intermediate and low, respectively. Finally, sections of OL1 and SC2 appeared to be more diverse. Intra-wetland comparisons indicated higher CDR values on the central transect in SC2, and more homogenized communities at the UP transect in OL1. Finally, MDI exhibited the highest CDR scores and elevated CD values.

Discussion

Sediment and Understory structure related to hydrological connectivity

Our results indicate that sediment structure at flooded areas of the selected wetlands was related to their hydroperiod. However, riparian understory structure corresponded to the specific wetland successional stage, which reflected the effects of hydrological connectivity over longer temporal and spatial scales than is considered for hydroperiod characterization.

As formulated by Mitsch and Gosselink (1993), the wetland hydroperiod concept is useful for comparing hydrological connectivity in floodplain wetlands. Others have described hydrological connectivity using structural variables, such as substrate grain size, or quantitatively, by measuring ecological processes, such as water residence time, water level changes, or water age (Van der Brink et al., 1993; Hein et al., 1999; Bornette et al., 2001; Roozen et al., 2003; Hein et al., 2004). Our results demonstrate that the Hysteretic Loop Method (floodplain water-level fluctuations correlated with river water level) is useful in classifying riparian wetlands according to their hydroperiod. An exact diagonal line in the plot indicates that the RWL of the site changed simultaneously with that of the river. The more circular the hysteretic loop, the longer the delay between the change in water level at the site and that of the river; a delay that reflects the dominance of river seepage inflow. Furthermore, the inflexion points during the rising phase of the flood highlight the importance of surface inputs at the point where river discharge reaches a specific threshold. In this study, four types of hydroperiod were identified: 1) Backflow channels permanently connected at their downstream end (BC1 and BC2); for this type, upstream inputs were of negligible importance during low magnitude flow. 2) Sites with loops controlled by a specific river-discharge threshold during the rising phase (OL1 and SC1); differences in hysteretic loops between these sites during the receding phase reflected the absence of upstream connection (OL1) versus the dominance of upstream inputs from a discharge threshold (SC1). 3) Groundwater-connected sites (AP and SC2); for these sites, rapid fluctuations in water level at the hyporheic zone promoted parallel changes at SC2 compared with those in the river. However, the more circular loop of AP reflected its greater distance from the river and the slow conductivity of sub-surface environments. 4) Disconnected riparian wetlands (OL2 and OL3); these oxbow lakes had a thick layer of fine sediment that impeded vertical connectivity during the examined flood.

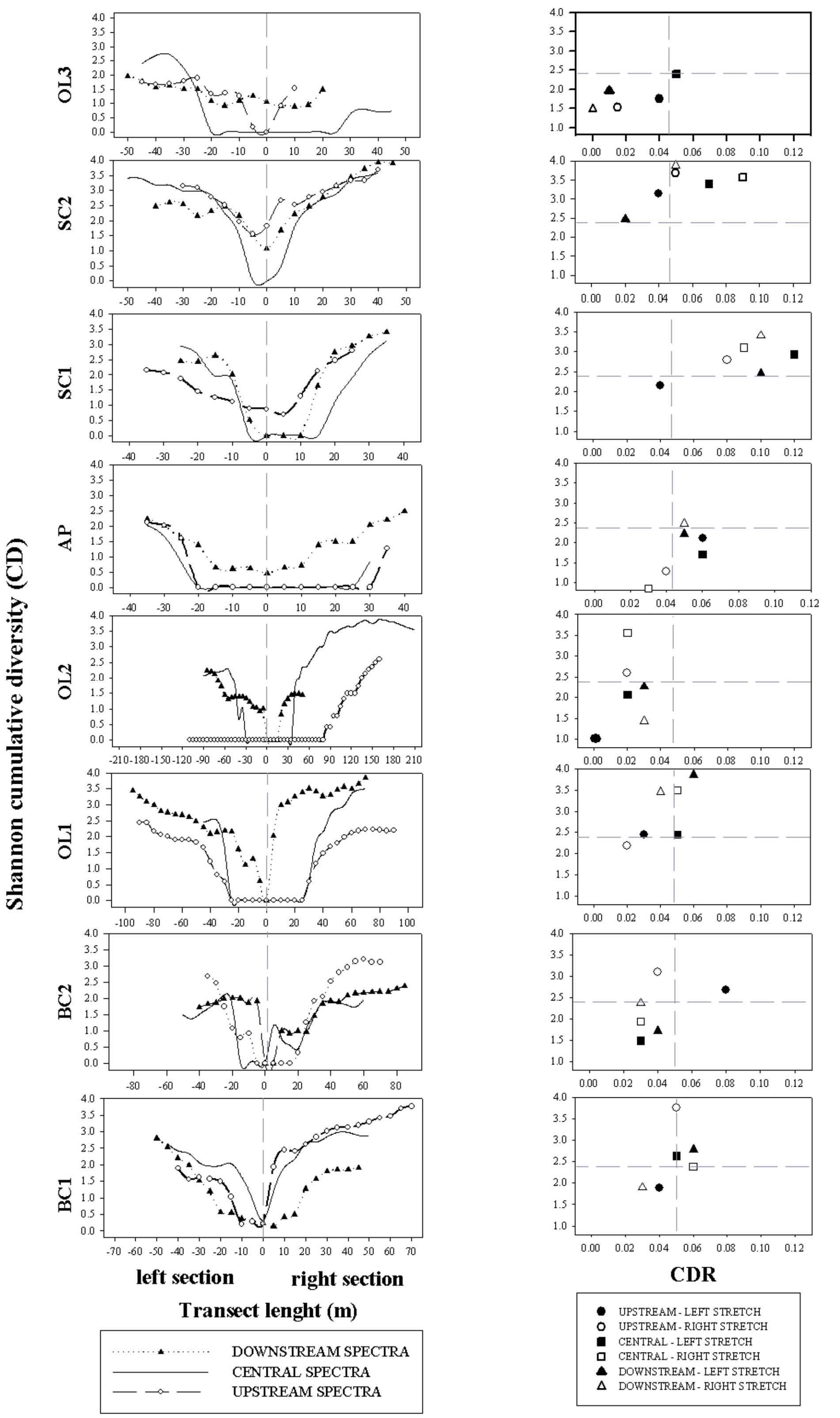

Figure 6. Cumulative Diversity Spectra and relationship between Cumulative Diversity (CD) and the Cumulative Diversity Rate (CDR). The longitude of the Transect Length Axis depends on the extent of the area considered to be Riparian Wetland at each location. Dashed lines indicate mean CD and CDR values for ease of inter-wetlands comparisons.

With regards to sediment structure, the results of the applied multivariate approach seemed consistent with the hysteretic loop evaluation. Tockner and Schiemer (1997), emphasized the importance of organic matter in sediments as an indicator of low surface connectivity, an idea first suggested by Rostan et al. (1987). The ORGSED location group included the sediments of the disconnected wetlands, OL3 and OL2 (Hydroperiod type 4). The substrate of a relatively old, accreted backwater channel, BC1 (see Fig. 5, Hydroperiod type 1), was also included in this group. These observations may indicate the positive effect of aging on carbon accumulation when water scouring and accretion rates are low, as has been reported in other natural and created wetlands (Schwarz et al., 1996; Mitsch et al., 2005). Conversely, MINSED and INTSED location groups coincided with the sediments of the remaining connected wetlands, SC1, SC2, BC1, BC2 and OL1 (Hydroperiod types 1, 2 and 3), where allogenous processes, such as the sedimentation of river material and the exchange of organic matter with the main channel, decreased the organic content of the sediment. Proximity to the main channel might have promoted greater accretion of more coarse and inorganic substrates, resulting in a lower affinity for major ions and higher hydraulic conductivity. This interpretation at least partially explains the differences in conductivity, bulk density and organic matter between INTSED and MINSED, the latter comprised of locations adjacent to the main channel. In contrast, BC2-D might have been included in the ORG cluster in winter because decreased connectivity favored internal processes in backwater systems (Heiler et al., 1995). In addition, the predominance of river seepage as the water source drives wetlands toward autogenic functioning (Phase II in Tockner et al., 1999a), as manifested by the inclusion of SC2-C (Hydroperiod Type 3) in the ORG group in winter.

Seasonal variability in river discharge had functional implications for substrates at the study locations. Recurrent floods prior to winter sampling created a gradient from MINSED sites, which had diluted environments with low ammonia/nitrate ratios, to ORGSED, where conductivity and organic matter were elevated and reduced nitrogen forms predominated. In summer, stagnant conditions caused conductivity to rise relative to values in winter (Tables 4,5). The resulting low oxygen levels and redox processing might explain, in part, the lower summer nitrate and sulphate content for ORGSED and INTSED locations. Elevated levels of nitrate, sulphate, and calcium in MINSED during this season probably resulted from greater oxygen availability and earlier salt deposition at four of its locations, which were not flooded and presented elevated values for factor 2 of the PCA (Fig. 4). Compounds dissolved in the interstitial water crystallize at the sediment surface and, consequently, gave rise to an accumulation of salts in the top layer, as occurs in freshwater marshes (Degroot et al., 1993).

Finally, wetlands successional stage significantly influenced the Cumulative Diversity (CD) Spectra (Fig. 6, left section). This influences wetland topography, thus controls the hydroperiod and associated characteristics such as duration and depth of flooding, variation in the water table, flood scouring, or soil grain-size, which are all associated with vegetation structure (Mountford and Chapman, 1993; Henry et al., 1994; David, 1996; Bornette et al., 1998; Leyer, 2004). The planar forms in central sections of the spectra indicate the dominance of highly connected plots, with the absence of vegetation or dominance by a single macrophyte species. Discrete or continuous environmental gradients are reflected in the rising phases of the spectra; diagonal forms indicate sequential gradients, and asymptotic forms correspond to step-wise gradients. At the selected wetlands, asymptotic shapes predominated due to the dominance of vertical accretion in the floodplain formation process. In some cases, natural trends are completely overridden by the human occupation of areas surrounding water bodies in the study area, as was observed for the left section of the upstream spectrum in OL2.

In areas adjacent to the main channel, sediment accretion progressively created such step gradients during initial wetland stages. Later, higher accretion rates in upstream areas smoothed those gradients in the central sections of the spectra, driving the transition from highly connected side channels to backflow channels, i.e., from lotic to more lentic conditions (Upstream Spectra in SC1, Fig. 6). This process occurred later in central and downstream areas, leading hydrological connectivity to decrease at the downstream end of these backflow channels, as observed in SC2. This same pattern probably occurs at longer intervals and on larger spatial scales in wetlands far from the main channel, such as oxbow lakes. Downstream zones in the wetland basin became accreted while shallow water or muddy areas prevailed in other sectors. Clearly, *Phragmites australis* and *Typha* sp. dominated plots in these other zones and, coupled with the absence of other macrophyte species, resulted in homogenous central sections (OL1, OL2 and OL3 in Fig. 6). Above the banks, mature forests, often dominated by *Populus alba*, held a more diverse understory (OL1 in Fig. 6) compared to the plots occupied by *Rubus ulmifolius*, as exemplified by the right section of the central section in OL3 (Fig. 6).

Ecological status of the study reach

Selected wetlands exhibited a wide range of hydroperiod types for these low-magnitude floods, results that may be taken as representative of the study reach since these wetlands account for 70% of the riverscape. However, we hypothesized that hydroperiod diversity would decrease during higher-magnitude events than the one used for this hydroperiod analysis (> 536 m^3/s, 0.14 y return period). The absence of channel migration since 1981 (Cabezas et al., unpublished data) has caused vertical accretion to dominate floodplain formation, and has restricted flooding to areas close to the main channel (“straitjacket” area *sensu* Lamers et al., 2006). At the study reach, the proportion occupied by riparian wetlands (3.9%) and the area of the floodplain that is permanently flooded (0.7%) are low. Consequently, the majority of the floodplain area is only flooded during high magnitude floods (> 2500 m^3/s, 5.36 y return period) and remains disconnected from the river most of the time. This means that almost no floods are capable of increasing the hydroperiod diversity of ancient wetlands. In highly connected wetlands inside the “straitjacket”, water sources will rapidly come to be dominated by superficial inputs, whereas connectivity thresholds will be reached sooner and will become less important. The prediction is that in side channels, hydroperiod types will become reduced to those observed in the backflow channels, BC1 and BC2, where water level fluctuations completely depended on river discharge variation. Disconnected wetlands, such as OL3 and OL2, have a thick sediment layer that impedes groundwater connectivity; superficial inputs only occur during low-frequency floods. Thomaz et al. (2007) emphasized that floods increase the similarity among aquatic habitats in river floodplain systems. In contrast, Malard et al. (2000) showed that the physico-chemical heterogeneity of a glacial riverscape was controlled by multiple water sources and flow-paths, and increased during expansion-contraction cycles. Similarly, the array of flooded sediments showed a considerable heterogeneity under the current flow regime (Fig. 5). However, flooded zones were scarce compared with the accreted areas, OW, RB and ORC (Table 1). Considering the entire reach, substrate homogeneity affects system functionality, for example, because processes occurring at low oxygen levels are delimited to small areas. These effects extend to reach species that depend on these habitats, leading to a decrease in reach biodiversity.

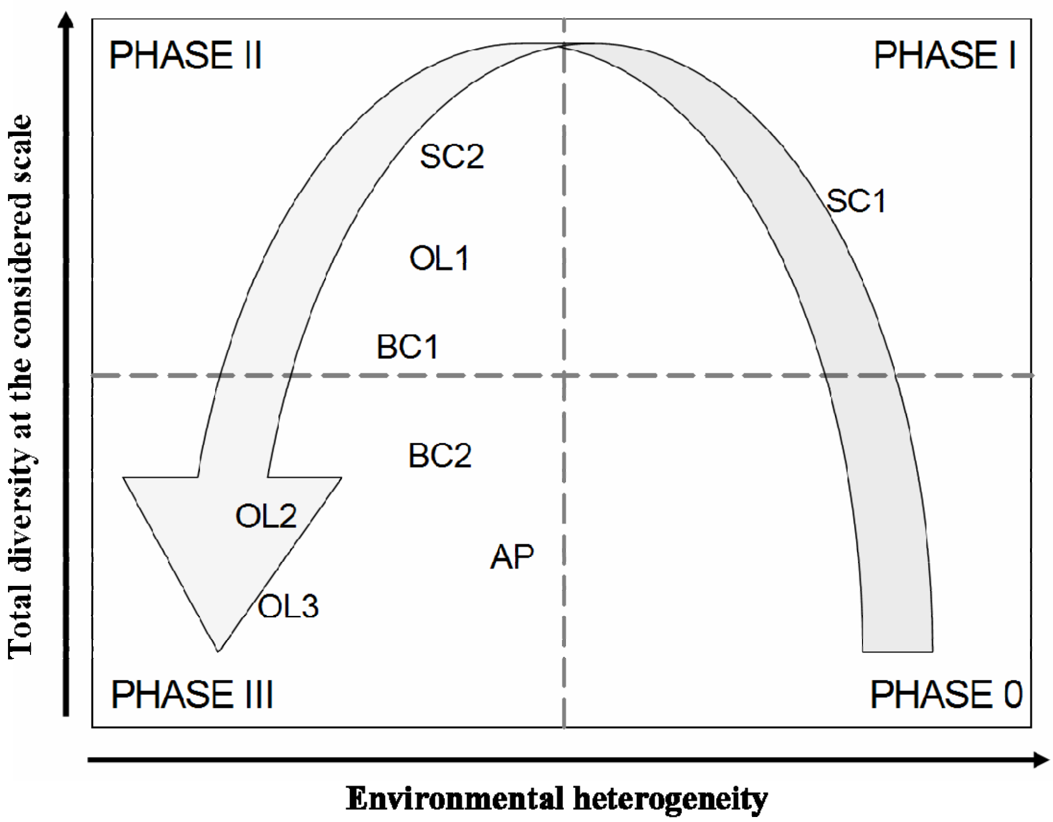

Figure 7. Two-dimensional conceptual model of wetland succession at the study reach. The successional stage of studied wetlands has been inferred from diversity measures at the section scale. Under actual conditions, the arrow indicates the unique direction of the succession
.

Using the understory analysis, a two-dimensional conceptual model (Fig. 7) was built to further assess the ecological status of the study reach. The convenience of functional measures of biodiversity has been emphasized in previous studies, which highlighted the importance of species turnover rate and overall diversity at the scale under consideration (Tockner et al., 1999b; Ward et al., 1999). The assignment of the examined wetlands to Phases II and III with low environmental heterogeneity stresses the fact that the riverscape at the study reach is homogenous and dominated by mature stages. Moreover, under current hydrogeomorphological conditions, the wetlands of the study reach will remain static or progress irreversibly to Phase III due to a lack of cyclic rejuvenation events. In our scheme, Phase I corresponds to wetlands where environmental conditions permit a heterogeneous distribution of plants along the section. The frequently connected backflow channel, SC1 (Table 1 and Fig. 6) represents this stage. Diverse, but spatially homogeneous, sections represent Phase II, which might develop as a consequence of intermediate levels of superficial connectivity or the influence of ruderal species. OL1, SC2 and BC1 represent this stage in the study reach. Phase III corresponds to a riparian area with marked, discrete gradients (permanently versus rarely flooded), which results in low diversity and heterogeneity. In our model, OL2, OL3, AP and BC2 are classified as Phase III. Open-water and reed plots dominated the deepest areas and brambles dominated the banks. Phase 0 probably occurs in areas that are potentially heterogeneous but where major counteracting factors exist. This might be the case in permanent side channels, where water availability is gradual, but there is a homogeneous stony substrate.

In conclusion, the effects of diminished river-floodplain interactions on the riverscape have clearly affected the ecological status of the study reach. We could classify the ecological status of the reach as deficient/acceptable (orange-yellow) following the Water Framework Directive terminology, although we have arrived at this assessment using completely different criteria. Biodiversity measurements showed a homogenous riverscape dominated by wetlands at mature successional stages. Moreover, the considerable hydroperiod and sediment diversity were found to be variable over longer spatio-temporal scales. Consequently, the lack of erosive floods within the reach seems to make inclusion of ecological restoration of geomorphological dynamics a highly appropriate management objective.

Implications for restoration and management

It appears that it would be unrealistic to expect current management policies of the Ebro basin to re-establish the geomorphological dynamics in the study reach. Different approaches appropriate to the scale of the objectives could be designed for the management of the Middle Ebro floodplains. Such alternative strategies would need to make more integrated use of natural resources, primarily soil and water. On the reach scale, it might be necessary to remove or redirect dikes to restore an ecosystem with the hydrogeomorphological capacity to maintain the evolving mosaic of habitats characteristic of a well-preserved floodplain (Buijse et al., 2002). Such changes would likely promote a more heterogeneous riverscape - one in which Phase 0 and 1 of our model would be represented in the reach. New wetlands would develop at concave banks due to channel migration, whereas succession would be truncated in other cases, driving current wetlands to younger stages. However, this self-sustaining approach must be accompanied by a lowering of the floodplain height because the high level of floodplain accretion will hamper main channel movement.

On the site scale, riverscape heterogeneity could be increased by creating artificial wetlands or modifying the local topography of existing wetlands. Phases 0 and 1 in our model would be represented at the study reach if fine sediments were removed from side channels, such as SC2 and SC1, located close to the main channel. For BC1 and BC2, lowering banks and upstream ends would encourage these sites to retreat towards previous phases. In the cases of the oxbow lakes, OL1, OL3 and OL1, it is probably not feasible to open the upstream ends because they are located quite far from the main channel. For these wetlands, a focus on improving downstream superficial connection and removing bottom sediments would increase sub-superficial connectivity, after assessing the effects. Water and sediment exchange with the river could be controlled through artificial systems to prevent rapid accretion and allow the hydroperiod to be controlled as desired. Banks should also be lowered and mature vegetation should be removed. On the accreted floodplain, artificial ponds could be created to restore Phase 3 wetlands to the study reach. The described approach would ensure a gradient of hydroperiod types during different magnitude floods, while notably increasing the permanently flooded area. However, this strategy is not self-sustaining unless the main channel migrates, so significant economic investments will be required to maintain periodic inputs of energy. These proposals are presented as a reasonable compromise between the need for flood protection and the desire for ecosystem rehabilitation in highly regulated rivers (Baptist et al., 2004).

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References

Aizpuru I, P.M Aseginolaza , P. Uribe-Echebarría , P. Urrutia and I. Zorrakin, 2003. Claves ilustradas de la Flora del País Vasco y territorios limítrofes. Servicio Central de Publicaciones del Gobierno Vasco - Eusko Jaurlaritzaren Argitalpen Zerbitzu Nagusia: Vitoria-Gasteiz, 831 pp.

Amoros, C. and G. Bornette, 2002. Connectivity and biocomplexity in waterbodies of riverine floodplains. Freshwater Biology **47**:761-776.

Baptist, M. J., W. E. Penning, H. Duel, A. J. M. Smits, G. W. Geerling, G. E. M. van der Lee and J. S. L. Van Alphen, 2004. Assessment of the effects of cyclic floodplain rejuvenation on flood levels and biodiversity along the Rhine river. River Research and Applications **20**:285-297

Bornette, G., C. Amoros and N. L. Lamouroux, 1998. Aquatic plant diversity in riverine wetlands: The role of connectivity. Freshwater Biology **39**:267-283.

Bornette, G., H. Piegay, A. Citterio, C. Amoros and V. Godreau, 2001. Aquatic plant diversity in four river floodplains: a comparison at two hierarchical levels. Biodiversity and Conservation **10**:1683-1701.

Brunke, M., 2002. Floodplains of a regulated southern alpine river (Brenno, Switzerland): ecological assessment and conservation options. Aquatic Conservation: Marine and Freshwater Ecosystems **12**:583-599.

Buijse, A. D., H. Coops, M. Staras, L. H. Jans, G. J. Van Geest, R. E. Grift, B. W. Ibelings, W. Oosterberg and F. C. J. M. Roozen, 2002. Restoration strategies for river floodplains along large lowland rivers in Europe. Freshwater Biology **47**:889-907.

Castro, P., J. Guerrero and M.A. Muñoz. 2001. Plan de Restauración del Bosque de Ribera en la Reserva Natural de los Galachos (Zaragoza). Consejo de Protección de la Naturaleza de Aragon, Zaragoza, 165 pp.

David, P. G., 1996. Changes in plant communities relative to hydrologic conditions in the Florida Everglades. Wetlands **16**:15-23.

Degroot, C. J. and C. Vanwijck, 1993. The impact of desiccation of a fresh-water Marsh (Garcines Nord, Camargue, France) on sediment water vegetation interactions .1. The sediment chemistry. Hydrobiologia **252**:83-94.

Galat, D. L., L. H. Fredrickson, D. D. Humburg, K. J. Bataille, J. R. Bodie, J. Dohrenwend, G. T. Gelwicks, J. E. Havel, D. L. Helmers, J. B.

Hooker, J. R. Jones, M. F. Knowlton, J. Kubisiak, J. Mazourek, A. C. McColpin, R. B. Renken and R. D. Semlitsch, 1998. Flooding to restore connectivity of regulated, large-river wetlands - Natural and controlled flooding as complementary processes along the lower Missouri River. Bioscience **48**:721-733.

Heiler, G., T. Hein, F. Schiemer and G. Bornette, 1995. Hydrological connectivity and flood pulses as the central aspects for the integrity of a river-floodplain system. Regulated Rivers-Research & Management **11**:351-361.

Hein, T., C. Baranyi, W. Reckendorfer and F. Schiemer, 2004. The impact of surface water exchange on the nutrient and particle dynamics in side-arms along the River Danube, Austria. Science of the Total Environment **328**:207-218.

Hein, T., G. Heiler, D. Pennetzdorfer, P. Riedler, M. Schagerl and F. Schiemer, 1999. The Danube restoration project: Functional aspects and planktonic productivity in the floodplain system. Regulated Rivers-Research & Management **15**:259-270.

Henry, C. P., G. Bornette and C. Amoros, 1994. Differential-Effects of Floods on the Aquatic Vegetation of Braided Channels of the Rhone River. Journal of the North American Benthological Society **13**:439-467.

Henry, C. P. and C. Amoros, 1995. Restoration ecology of riverine wetlands .1. A scientific base. Environmental Management **19**:891-902.

Hughes, F. M. R., W. M. Adams, E. Muller, C. Nilsson, K. S. Richards, N. Barsoum, H. Decamps, R. Foussadier, J. Girel, H. Guilloy, A. Hayes, M. Johansson, L. Lambs, G. Pautou, J. L. Peiry, M. Perrow, F. Vautier and M. Winfield, 2001. The importance of different scale processes for the restoration of floodplain woodlands. Regulated Rivers-Research & Management **17**:325-345.

Junk, W. J., B. Bayley and R. E. Sparks, 1989. The flood pulse concept in river-floodplain systems. Special Issue of J.Canadian Fisheries and Aquatic Sciences. **106**:11-27.

Kondolf, G. M., 1998. Lessons learned from river restoration projects in California. Aquatic Conservation: Marine and Freshwater Ecosystems **8**:39-52.

Lamers, L. P. M., R. Loeb, A. M. Antheunisse, M. Miletto, E. C. H. E. T. Lucassen, A. W. Boxman, A. J. P. Smolders and J. G. M. Roelofs, 2006. Biogeochemical constraints on the ecological rehabilitation of wetland vegetation in river floodplains. Hydrobiologia **565**:165-186.

Leyer, I., 2004. Effects of dykes on plant species composition in a large lowland river floodplain. River Research and Applications **20**:813-827.

Malard, F., K. Tockner and J. V. Ward, 2000. Physico-chemical heterogeneity in a glacial riverscape. Landscape Ecology **15**:679-695.

Metzger, J. P. and H. Décamps, 1997. The structural connectivity treshold : an hypothesis in conservation biology at landscape scale. Acta Oecologica **18:**1-12.

Mitsch W.J. and J.G. Gosselink . 1993. Wetlands, Van Nostrand Reinhold, New York, 722 pp.

Mitsch, W. J., L. Zhang, C. J. Anderson, A. E. Altor and M. E. Hernandez, 2005. Creating riverine wetlands: Ecological succession, nutrient retention, and pulsing effects. Ecological Engineering **25**:510-527.

Mountford, J. O. and J. M. Chapman, 1993. Water regime requirements of british wetland vegetation - using the moisture classifications of Ellenberg and Londo. Journal of Environmental Management **38**:275-288.

Ollero, A., 2007. Channel adjustments, floodplain changes and riparian ecosystems of the middle Ebro River: Assessment and management. International Journal of Water Resources Development **23** 1:73-90.

Regato, P., 1988. Contribucion al estudio de la flora y la vegetación del galacho de la Alfranca en relación con la evolución del sistema fluvial, Diputación General de Aragon, Zaragoza, 189 pp.

Roozen, F. C. J. M., G. J. Van Geest, B. W. Ibelings, R. Roijackers, M. Scheffer and A. D. Buijse, 2003. Lake age and water level affect the turbidity of floodplain lakes along the lower Rhine. Freshwater Biology **48**:519-531.

Rostan, J. C., C. Amoros and J. Juget, 1987. The organic content of the surficial sediment - a method for the study of ecosystems development in abandoned river channels. Hydrobiologia **148**:45-62.

Schwarz, W. L., G. P. Malanson and F. H. Weirich, 1996. Effect of landscape position on the sediment chemistry of abandoned-channel wetlands. Landscape Ecology **11:**27-38.

Sparks, R. E., 1995. Need for ecosystem management of large rivers and their floodplains. Bioscience **45**:168-182.

Tabacchi, E., D. L. Correll, R. Hauer, G. Pinay, A. M. Planty-Tabacchi and R. C. Wissmar, 1998. Development, maintenance and role of riparian vegetation in the river landscape. Freshwater Biology **40**:497-516.

Thomaz, S. M., L. M. Bini and R. L. Bozelli, 2007. Floods increase similarity among aquatic habitats in river-floodplain systems. Hydrobiologia **579:**1-13.

Tockner, K., F. Malard and J. V. Ward, 2000. An extension of the flood pulse concept. Hydrological Processes **14**:2861-2883.

Tockner, K., D. Pennetzdorfer, N. Reiner, F. Schiemer and J. V. Ward, 1999a. Hydrological connectivity, and the exchange of organic matter and nutrients in a dynamic river-floodplain system (Danube, Austria). Freshwater Biology **41**:521-535.

Tockner, K. and F. Schiemer, 1997. Ecological aspects of the restoration strategy for a river-floodplain system on the Danube River in Austria. Global Ecology and Biogeography Letters **6**:321-329.

Tockner, K., F. Schiemer, C. Baumgartner, G. Kum, E. Weigand, I. Zweimuller and J. V. Ward, 1999b. The Danube restoration project: Species diversity patterns across connectivity gradients in the floodplain system. Regulated Rivers-Research & Management **15**:245-258.

Tockner, K., F. Schiemer and J. V. Ward, 1998. Conservation by restoration: The management concept for a river-floodplain system on the Danube River in Austria. Aquatic Conservation-Marine and Freshwater Ecosystems **8**:71-86.

Tockner, K. and J. A. Stanford, 2002. Riverine floodplains: present state and future trends. Environmental Conservation **29**:308-330.

Vandenbrink, F. W. B., J. P. H. M. Deleeuw, G. Vandervelde and G. M. Verheggen, 1993. Impact of hydrology on the chemistry and phytoplankton development in floodplain lakes along the lower Rhine and Meuse. Biogeochemistry **19**:103-128.

Ward, J. V., 1989. The 4-dimensional nature of lotic ecosystems. Journal of the North American Benthological Society **8**:2-8.

Ward, J. V., 1998. Riverine landscapes: Biodiversity patterns, disturbance regimes, and aquatic conservation. Biological Conservation **83**:269-278.

Ward, J. V. and J. A. Stanford, 1995. Ecological connectivity in alluvial river ecosystems and its disruption by flow regulation. Regulated Rivers-Research & Management **11**:105-119.

Ward, J. V., K. Tockner and F. Schiemer, 1999. Biodiversity of floodplain river ecosystems: Ecotones and connectivity. Regulated Rivers-Research & Management **15**:125-139.

Ward, J. V., K. Tockner, U. Uehlinger and F. Malard, 2001. Understanding natural patterns and processes in river corridors as the basis for effective river restoration. Regulated Rivers-Research & Management.

STUDY IV

Carbon and nitrogen accretion in the topsoil of the Middle Ebro River Floodplains (NE Spain): Implications for their ecological restoration

Carbon and nitrogen accretion in the topsoil of the Middle Ebro River Floodplains (NE Spain): Implications for their ecological restoration

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Abstract

This study aimed at evaluating the potential of floodplains in the Middle Ebro River (Spain) to accumulate organic carbon and nitrogen through sedimentation. Total organic carbon (TOC) and nitrogen (TN) in the top soil of riparian habitats and in the river sediments of one river reach were examined and related to landform and land use to gain insight in the current patterns of soil organic matter accumulation. Based on our results, the potential of floodplain soils as OM sinks was assessed to propose a rehabilitation framework which includes carbon and nitrogen accumulation. To achieve those goals, six categories of landform evolution, including crops and poplar groves, were defined using aerial photographs. The study plots (n=18), one per category, were set in three areas of the study reach. SOM quantity (TOC, TN) and quality (C:N, Non Hydrolizable Carbon) were characterized for each plot. With respect to the river sediments, the material deposited after one flood was analyzed to estimate the relationship between grain size and organic matter content. It was used to infer the relative importance of allochthonous vs. autochthonous OM inputs in the top soil of the study plots. According to our results, landform evolution influences the quantity but not the quality of top soil OM in the Middle Ebro floodplains. Natural patches > 60 yr old incorporated *in-situ* produced organic matter and presented the highest OC and N stocks. In turn, sedimentation was the dominant process in SOM dynamics at younger natural patches. Furthermore, approximately half of the OC could be included within the passive pool. In any case, anthropogenic land use counteracts the ability of floodplain soils to act as nitrogen and carbon sinks; thus, the rehabilitation of the floodplain towards natural land-covers is required.

Keywords: Organic carbon, nitrogen, recalcitrant carbon, riparian soils, Ebro, floodplains, sedimentation.

Introduction

The major role of riverine floodplains as sinks, sources or transformers of organic matter (OM) has been highlighted by many researchers (Junk et al. 1989; Ward and Stanford 1995; Walling et al. 1996; Tockner et al. 1999; Robertson et al. 1999; Hein et al. 2003). However, studies that address the role of a floodplain as a permanent OM sink are scarce, despite the importance of organic carbon and nitrogen accumulation in these systems. In riparian ecosystems, nitrogen retention can improve the quality of surface water (Johnston 1991; Mitsch et al. 2000; Mitsch et al. 2001; Day et al. 2004; Mitsch et al. 2005a; Verhoeven et al. 2006) and thereby prevent the eutrophication of downstream aquatic ecosystems. In terms of carbon retention, increasing the ability of ecosystems to act as carbon sinks has been suggested as a measure for offsetting the greenhouse effect (IPCC 2000, Mitsch et al. 2005b). The erosion and redistribution of carbon within the watershed might have a major influence on C dynamics because riparian wetlands can support long-term budgets (McCarthy and Ritchie, 2002). Soil organic matter (SOM) has a special relevance in this context since it forms a larger and more lasting pool than living biomass. SOM is also a complex pool consisting of various distinct kinds of residues at different stages of decomposition. Understanding this complexity is important for a proper evaluation of the influence of OM quality (i.e., degradability, Rovira et al. 2008) on the potential of floodplains to act as OM sinks.

To evaluate C and N accretion in floodplain substrates, it is necessary to have a clear understanding of the effect of processes operating at different spatio-temporal scales. At the watershed scale, riverine floodplains can buffer rivers from the washload produced in the upstream parts of the catchment (Asselman and Middelkoop, 1995; Steiger and Gurnell, 2003; Noe and Hupp, 2005). At the landform scale, OM produced by biotic assemblages is incorporated into surface soils in the intervals between floods. During floods, low-organic fluvial sediment may be deposited on riparian landforms, and OM stored in different compartments of the system (i.e., aboveground biomass, sediment and litter) is either exported or buried. As noted by Daniels (2003), the accretion rate is inversely related to soil development. The driving force that determines the relative importance of either autochthonous or allochthonous OM inputs is hydrological connectivity, which is a measure of the exchange of mass and energy between the river and its floodplain across different spatio-temporal scales (Junk et al. 1989; Tockner et al. 2000; Amoros and Bornette 2002).

Under natural conditions, early created landforms are covered by mature stages of riparian forest, which are flooded only during low-frequency events (Amoros and Wade, 1996). Contrastingly, hydrological connectivity is higher in recent landforms occupied by early successional stages. However, the transformation of riparian forests into agricultural fields or poplar groves reduces the incorporation to their substrates of either autochthonous OM inputs, since they are harvested, or allochthonous inputs, since those patches are protected by dykes. By using aerial pictures, we analysed landform age and evolution in order to evaluate the influence of hydrological connectivity, i.e. dominance either autochthonous or allochthonous OM inputs, on SOM accretion. Since the upper 10 cm were sampled, we selected this approach to infer hydrological conditions for a period long enough to be related with SOM.

At the Middle Ebro floodplains, most of the area is subject to anthropogenic land-uses (Ollero, 1992). The natural flow regime has been greatly modified during the last century (Comin, 1999), and the exchange of material between the river and the adjacent landforms has been disrupted by dykes. Even so, there exists a heterogeneous matrix of patches with different geomorphic evolution and connectivity type. This spatial landscape setting allowed us to define the following objectives: (1) to evaluate the influence of landform evolution on the accumulation of soil organic matter (SOM) in the topsoil of natural riparian ecosystems; (2) to examine, within a floodplain perspective, the role of riparian soils as SOC and SON sinks; (3) to propose a valid management and rehabilitation framework that includes SOC and SON accretion as restoration targets.

Study area

The Ebro River is the largest river in Spain (910 km long, has a watershed of 85,362 km^2, and an average annual

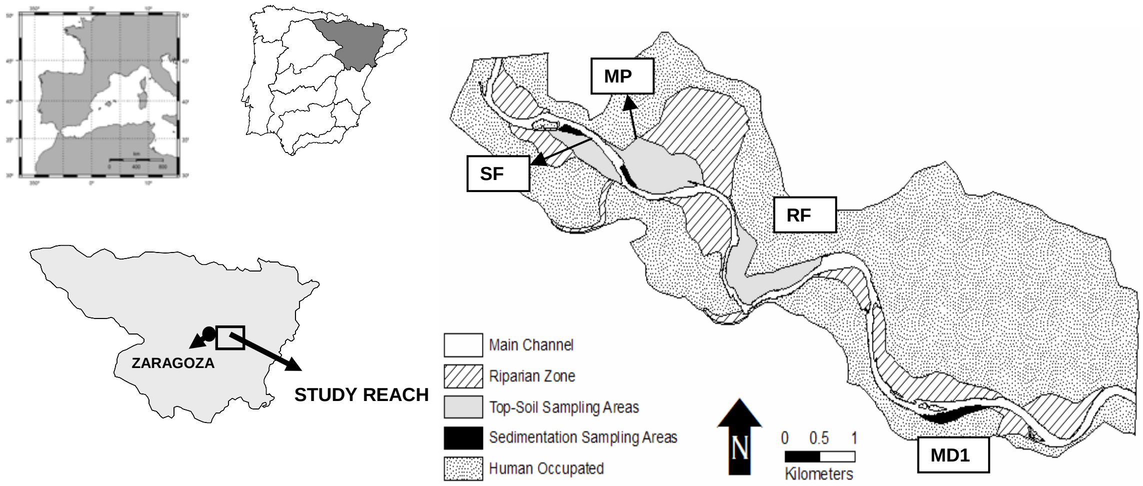

Figure 1. Location of the study area, which was delimited by the 25-year floodplain, and the riparian areas where topsoil and sedimentation samples were collected along a reach on the Middle Ebro River Floodplain, Spain. RF = Rincon Falso; MP = Mejana de Pastriz; SF = Soto Francés; MD1 = Margen Derecha

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discharge into the Mediterranean Sea of 14,442 hm^3; 1927-2003). The study area (Fig. 1) was a reach on the Middle Ebro River, 12 km downstream of Zaragoza, NE Spain. That section of the Ebro River meanders (sinuosity = 1.39, slope of the bankfull channel = 0.050%) and, on average, the floodplain is about 5 km wide (Ollero 1995). At the study area, the average monthly discharge is 230 m^3/s (CEDEX 1997) and the elevation ranges from 175 m a.s.l. in the river channel to 185 m a.s.l. at the base of the scarp. In the last century, the water discharge of the Ebro River had declined steadily because of an increase in the diversion of water, primarily, for agricultural irrigation and retention in reservoirs within the Ebro River watershed (Comín, 1999). In that period, dykes were built to protect against floods and the majority of the floodplain was used for agriculture. Between 1981 and 2007, the main channel did not migrate. Consequently, the connectivity between the river and the floodplain has been drastically reduced, which has led to the homogenization and terrestrialization of the riverscape (Cabezas et al, in press), where vertical accretion was the dominant process in the development of the floodplain.

Methods

Topsoil Sampling

The landscape dynamics at the study area was inferred from black-and-white orthoimages taken in 1927, 1956, 1981, and 1998. For each year, geomorphological maps were digitized and patches were classified as either: 1) Main Channel; 2) Agricultural; 3) Poplar Grove; 4) Wetland; 5) Intermediate, patches which had vegetation at intermediate stages of succession; 6) Mature Forest, patches which contained developed mature forests with older trees; 7) Gravel/Coarse Sediment (frequently flooded areas mostly covered by coarse gravel and scarce vegetation). An age map of the current landforms was then generated using the entire set of aerial pictures. The data from 1956 and 1998 were used to create a transition map, and patches were assigned to one of the following six categories: (1) *Highly Flooded (HF)* included young and frequently flooded areas classified as Gravel/Coarse Sediment in 1956 and 1998, although in 1956 they also occurred in the main channel; (2) *Mature Forest (MF)* included patches that were classified as "Intermediate" in 1956 and "Mature" in 1998; (3) *Intermediate (INT)* included areas classified as "Main Channel" or "Gravel/Coarse Sediment" in 1956, but in 1998 were occupied by intermediate stages of riparian forests; (4) *Connected Forest (CF)* included areas that had a history similar to those in INT, but are flooded during ordinary flood events; (5) *Agricultural (AGR)* and (6) *Poplar (POP)* included areas with crops and *Populus sp*. Groves, respectively. Digitizing and geoprocessing were performed using ArcMap 8.3® .

Three natural riparian zones, *Mejana de Pastriz* (MP), *Rincón Falso* (RF) and *Soto del Francés* (SF) (Fig. 1) were chosen because they encompassed all of the "natural" *(MF, INT, CF, HF)* sampling categories. In each area, the patches assigned to those categories have developed uniquely since 1956 (Fig. 2). The changes in their distribution among categories might mean that there are differences in their substrates. In each zone, adjacent agricultural fields and poplar groves were selected randomly. To identify the zones of homogeneous vegetation inside the patches, a field survey was performed. Patches were selected and designated as sampling plots. The *HF* plots in MP and RF were located in areas that had fine sediment, but were surrounded by larger areas of gravel and cobbles. In each of the plots, topsoil samples (litter layer not included, depth = 0-10 cm, n = 206) were collected using an undisturbed soil sampler (∅ = 5 cm, P1.31 Eijkelkamp®) along transects that covered the entire area of the plot (Fig. 2, Tab. 1), except in the HF in *Rincón Falso*. By using transects, a grid sampling strategy was achieved. It was chosen to reflect the spatial gradients, mainly related with flow paths, which can affect OM accretion. The number of samples collected in each riparian zone (Tab. 2, Fig. 2) varied proportionally to their size to get a similar detail for all the plots within the same category. Location of sampling points along the transects was planned to be equidistant, although the final position was slightly modified for some of the points to represent the dominant features (in terms of vegetation) of their surroundings.

In order to determine the dry mass and percentage moisture by mass on a wet-weight basis, 10-g sub-samples of fresh soil were oven-dried at 105 ºC. Bulk density was calculated as the dry mass per unit volume (g/cm^3). Samples were air-dried, weighed, and passed through a 2-mm sieve. Total Organic Carbon (TOC) and Total Nitrogen (TN) were measured using elemental analysis (Leco SC-144DR® and Elementar Variomax CN®, respectively). The quality of the Organic Carbon was estimated using single-step acid hydrolysis (Rovira and Vallejo, 2000) and the results were used to calculate the Refractory Index for Carbon (RIC), which is expressed as the proportion (%) of non-hydrolyzed organic carbon (NHC). Estimates of Recalcitrant Organic Carbon (ROC) were calculated by multiplying RIC by TOC, and C:N ratios were calculated by dividing TOC by TN. Grain-size analysis was performed using a laser-diffraction instrument (Coulter LS 230, Beckman Coulter®). Based on the data from 1927, 1956, 1981, and 1998, plot age was calculated as the mean age (yr) since

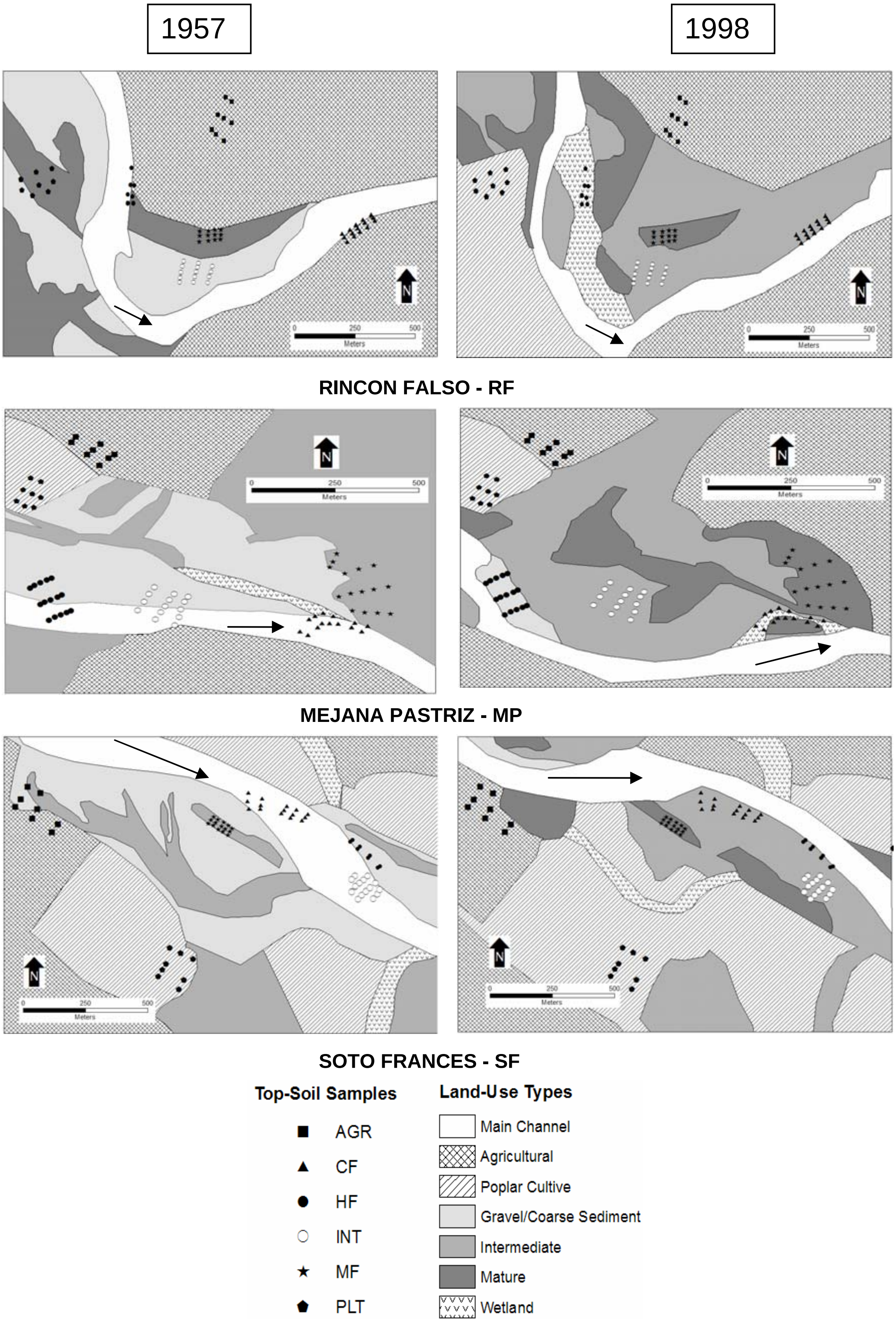

Figure 2. Locations of the topsoil sampling sites along transects in three areas of the study site within the Middle Ebro River Floodplain, Spain, which are overlain on land-use maps derived from 1957 and 1998 orthoimages taken in 1957 and 1998. Arrows indicate the direction of the flow of the Ebro River. HF= Highly Flooded; INT= Intermediate; CF= Connected Forest; MF= Mature Forest; AGR= Agriculture; POP= Poplar Grove.

PLOT DESCRIPTION					MEAN ± STANDAR ERROR						
Label		**N**	**Age**	**SC**	**Silt plus Clay**	**BD**	**TOC**	**TN**	**C:N**	**RIC**	**ROC**
			yr	**m^3/s**	**%**	**g cm^{-3}**	**g C m^{-3}**	**g N m^{-3}**		**%**	**g C m^{-2}**
MP	**HF**	15	17.5	300	39.58±6.54^{a}	1.20±0.03^{b}	1573.3±243.66^{a}	135.1±20.77^{a}	14.33±1.93^{d}	47.74 ±4.72^{a,b}	720.6±114.40^{a}
	INT	15	38.5	800	75.80±2.04^{c}	1.17±0.02^{b}	2141.4±73.22^{b}	185.4 ±5.75^{a,b}	11.56±0.18^{b}	48.87±0.91^{b}	1043.2±32.62^{a}
	CF	12	38.5	650	88.06±2.06^{d}	1.03±0.09^{a,b}	1903.3±187.23^{a,b}	175.5±16.94^{a,b}	10.93±0.29^{b}	49.43±0.70^{b}	944.0±92.67^{a}
	MF	15	68.5	1000	91.76±1.04^{d}	0.99±0.03^{a}	3831.8±101.61^{c}	310.4±7.09^{c}	12.34±0.12^{c}	53.10±1.04^{c}	2036.3±71.43^{b}
	AGR	8	68.5	2000	54.56±5.00^{b}	1.41±0.03^{c}	1933.3±114.56^{a,b}	211.1±6.93^{b}	9.22±0.56^{a}	43.39±0.91^{a}	842.8±58.96^{a}
	POP	8	68.5	2000	71.80±3.75^{c}	1.17±0.03^{b}	2010.2±188.96^{a,b}	171.3±12.00^{a,b}	11.64±0.45^{b}	51.14±2.31^{b}	1002.7±60.58^{a}
RF	**HF**	7	17.5	300	66.24±11.34^{a}	1.27±0.05^{c}	1547.3±232.01^{a}	153.5±24.55^{a}	10.42±0.47^{a}	54.78±1.50^{c}	839.2±118.20^{a}
	INT	14	38.5	800	81.87±1.81^{a,b}	1.19±0.02^{b,c}	2123.6±124.51^{a}	177.3±9.84^{a}	12.03±0.33^{b}	48.69±0.83^{a,b}	1035.3±64.71^{a}
	CF	15	38.5	600	76.30±3.70^{a}	1.13±0.03^{a,b}	2022.4±131.07^{a}	189.0±10.56^{a}	10.68±0.23^{a}	53.68±0.80^{c}	1086.6±73.76^{a}
	MF	13	> 68.5	1000	92.46±1.07^{b}	1.05±0.03^{a}	3015.4±104.80^{b}	254.8±8.41^{b}	11.84±0.16^{b}	51.73±1.32^{b,c}	1562.2±68.76^{b}
	AGR	8	68.5	2000	71.27±3.38^{a}	1.30±0.05^{c}	1715.9±181.45^{a}	180.3±15.90^{a}	9.49±0.67^{a}	45.96±3.05^{a}	780.5±105.62^{a}
	POP	9	38.5	3000	68.76±3.50^{a}	1.27±0.02^{c}	1632.3±82.65^{a}	161.0±10.01^{a}	10.20±0.20^{a}	55.09±0.98^{c}	894.6±35.22^{a}
SF	**HF**	8	17.5	200	86.86±7.74^{b}	1.17±0.05^{a,b}	1882.1±227.78^{b}	181.8±20.78^{a}	10.39±0.35^{a,b}	49.08±1.50^{a}	918.5±227.78^{a,b}
	INT	15	38.5	950	95.05±0.73^{b}	1.17±0.02^{a,b}	2771.7±106.43^{c}	245.5±7.57^{b}	11.26±0.16^{a,b}	49.43±0.84^{a}	1376.6±106.43^{c}
	CF	14	38.5	700	68.30±6.42^{a}	1.18±0.02^{a,b}	1779.1±206.51^{b}	140.6±15.82^{a}	13.10±1.62^{a,b}	48.40±2.96^{a}	815.7±206.51^{a,b}
	MF	15	68.5	950	73.37±2.97^{a}	1.07±0.02^{a}	3131.1±161.20^{c}	234.6±7.41^{b}	13.34±0.52^{b}	51.47±1.65^{a}	1592.7±161.20^{c}
	AGR	7	68.5*	2000	58.03±3.20^{a}	1.35±0.03^{c}	996.1±107.72^{a}	136.7±6.82^{a}	7.41±0.76^{a}	58.79±4.21^{b}	645.6±107.72^{a}
	POP	8	68.5	1500	67.44±2.91^{a}	1.28±0.04^{b,c}	1943.8±98.99^{b}	171.0±12.10^{a}	11.51±0.36^{a,b}	52.92±1.63^{a}	1026.7±98.99^{b}

Table 2. Summary results for the analyzed topsoil variables grouped by sampling plot. All variables presented significant differences for comparisons within each study area. Small print indicates the sub-groups formed after the applied post-hoc comparisons. MP= Mejana de Pastriz; RF= Rincon Falso; SF= Soto del Francés; HF= Highly Flooded; INT= Intermediate; CF= Connected Forest; MF= Mature Forest; AGR= Agriculture; POP= Poplar Grove.

	Bulk Density	TOC	TN	C/N	RIC	ROC
	p value	p value	p value	p value	p value	p value
Corrected Model	0.442	0.000**	0.000**	0.170	0.357	0.000**
Fine	0.214	0.000**	0.000**	0.873	0.481	0.000**
Riparian Area	0.268	0.027*	0.039*	0.269	0.161	0.140
Category	0.228	0.000**	0.000**	0.677	0.278	0.000**
Riparian Area*Category	0.571	0.001**	0.000**	0.260	0.371	0.000**
Corrected R^2	0.216	0.664	0.704	0.030	0.008	0.700
Lack of Fit	p value	p value	p value	p value	p value	p value
	0.626	0.997	0.544	0.099	0.607	0.928

Table 1. Results of an ANCOVA test using samples from "natural" plots (HF, CF, MF, and INT) within the study area on the Middle Ebro River Floodplain, Spain.. TOC= Total Organic Carbon; TN= Total Nitrogen; C:N= Carbon/Nitrogen ; RIC= Recalcitrant Index for Carbon; ROC= Recalcitrant Organic Carbon. * =p<0.05; ** = p<0.01.

channel abandonment. After ensuring that the data met the assumption of normality, (including transformations where appropriate), a one-way ANOVA was performed using SPSS® 14.0 to test for significant differences between categories in each of the riparian areas. Depending on the homogeneity of the variance, either SNK or Tahmane Tests were used in post-hoc comparisons. Differences in C:N ratios and RIC in the MP were evaluated using non-parametric tests (Kruskal-Wallis and Mann-Whitney Tests). To detect the effect of natural landform evolution on OM quantity and quality, Bulk Density, RIC, ROC, TN, TOC, and C:N ratio from natural sampling categories (*MF*, *HF*, *CF*, *INT*), were subjected to an analysis of covariance (ANCOVA), with Plot Category and Riparian Area as fixed factors, and Silt plus clay (% particles < 63 μm) as the covariable, using the GLM procedure in SPSS® 14.0.

Sedimentation

Sediment traps, made of artificial grass mats, were used to collect the sediment deposited by a single flood (8 days) on 1 January 2006, which reached 754,44 m^3/s (0.23 y, 1927-2003) at the Zaragoza gauge station (12 km upstream from the study area). The sediment traps were placed in three zones that had distinct microtopographies: *Mejana de Pastriz* (MP), *Soto del Francés* (SF), and *Margen Derecha* (MD). In each zone, 11 1-m^2 plots were established on transects that ran perpendicular to the main channel, although some of the plots were moved, not more than 5 m away from the original location, to avoid abrupt topographic changes and accumulations of woody debris (Fig. 3). The shape and size of each area dictated the space between transects and between plots. In each plot, three 25*25-cm sediment traps were affixed to the surface using 14-cm steel pins. A few days after the flood event, when all of the mats had re-emerged, they were taken to the lab and oven-dried at 60 ºC. Only 7 % of the artificial grass mats were flushed away by the river. To estimate sedimentation rates, all of the sediments were removed from the mats by hand, using a brush that had metallic bristles, and weighed. Calculations of TOC and TN, and the Grain-Size Analysis were performed as described above for the topsoil samples. The significance of the relationships between silt plus clay (% particles < 63 μm), TOC, and TN were explored due to the influence of grain size on carbon

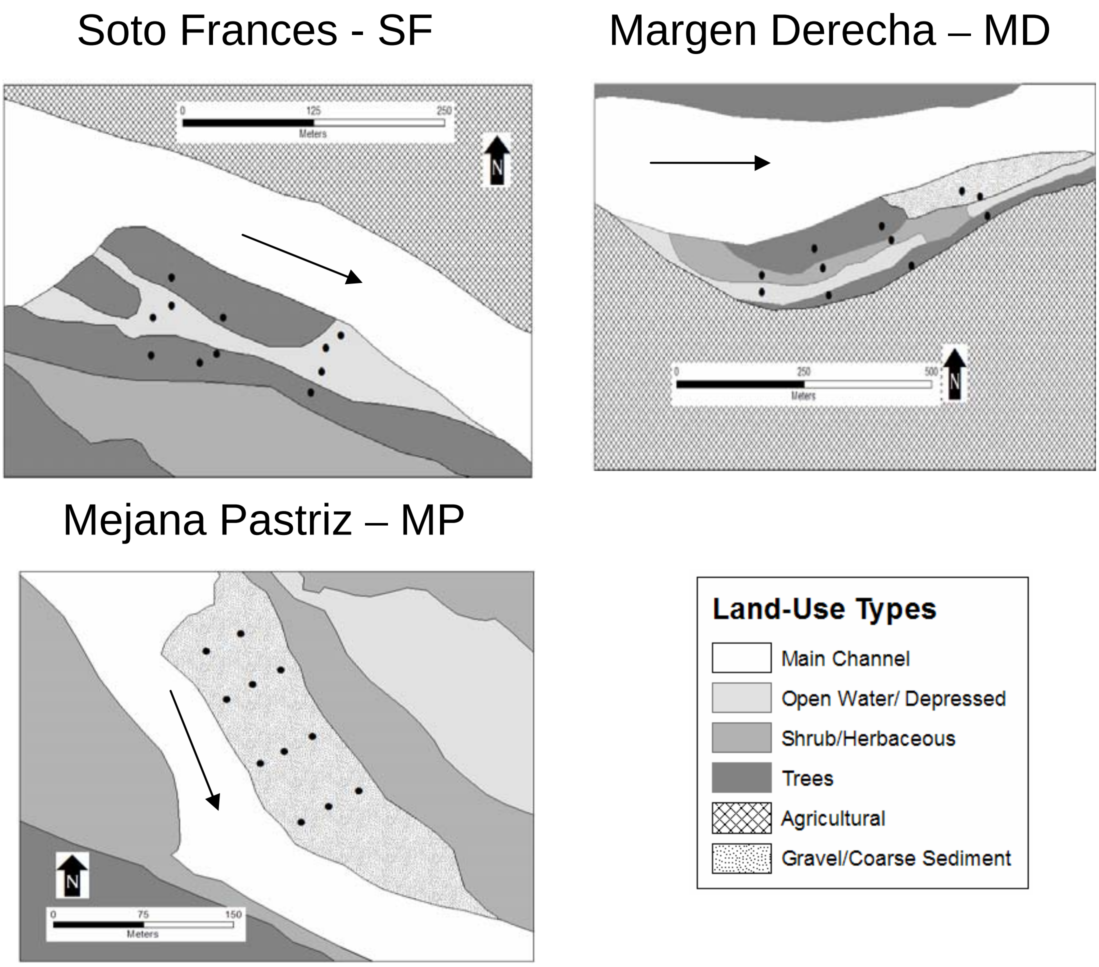

Figure 3. Locations of the sedimentation sampling sites in three areas of the study site within the Middle Ebro River Floodplain, Spain, which are overlain on land-use maps derived from 1998 orthoimages. Arrows indicate the direction of the flow of the Ebro River.

.and nitrogen deposition during floods (Asselman and Middelkoop, 1995; Walling and He, 1997; Steiger and Gurnell, 2003). Descriptive statistics were examined using SPSS® 14.0.

Incorporation into the topsoil of organic matter produced in situ

The silt plus clay fraction in top soil samples served as the independent variable to estimate, using the characterization of river sediments explained in the previous section, the theoretical TOC and TN contents when the material was deposited (Initial hereafter). Therefore, we assume that all the material contained in top soil samples had been previously deposited during floods, and, that the TOC and TN initial concentrations, in this material, depends on its grain size composition (Asselman and Middelkoop, 1995; Walling and He, 1997; Steiger and Gurnell, 2003). Initial and Observed (Measured in the top soil samples) TOC and TN values were then plotted together separating samples by plot age, which had been estimated as an average age since landform formation using the set of aerial pictures.

Results

Top soil characteristics

All of the factors included in the ANCOVA were significant to explain TOC and TN variability (p-value of corrected model in Tab. 1). Indeed, the corrected r^2 values, which reflect the adjustment to a linear relationship between the observed and predicted values, indicate that 70% of the variability in TOC and TN were explained by the effects of the individual factors, their interactions, and the covariable, which were statistically significant for TOC and TN (Tab.1). Neither collectively nor individually did the factors evaluated explain a significant amount of the variance in organic matter quality, as indicated by ANCOVA results for RIC and C:N (Tab. 1). ROC content was more strongly influenced by the quantity of SOC than by the biochemical complexity, since ANCOVA results were similar to those for TOC. The GLM explained only 20% of the variance in bulk density and, individually, none of the factors explained a significant amount of the variance (Tab. 1). That said, all factors fit the statistical model adequately for all of the variables examined because the opposite hypothesis was discarded because of p-values in the lack of fit test (Tab. 1)

Significant differences between sampling categories were observed for all variables in the three riparian zones (Tab.2). In *Mejana de Pastriz* (MP), the content of fine particles was the highest in *MF* and *CF*, whereas *AGR* and *HF* had coarser particle fractions. TOC and TN values were highest in the *MF* and lowest in the *HF*. The *HF* sites showed the greatest variability in the C:N ratio and the RIC, which was highest in the *MF* and *POP* and lowest in the *AGR*. Differences in ROC were similar to those in TOC with the exception of AGR.

At *Rincón Falso* (RF), however, variability in the data attributed to plot category was different from that observed in *MP* (Tab. 2). Although the finest substrate was found in the *MF*, the values of the other plot categories were about 70%, with the exception of INT (81%). Mature vegetation stages (*MF*) had the highest amounts of TOC and TN, although the difference between these sites and the *INT* and *CF* were less than they were in *Mejana de Pastriz*. C:N ratios were highest in the *MF* and *INT* sites and lowest in the *AGR* sites. Nevertheless, proportions of RIC were higher at *HF*, *CF*, and *POP* sites than they were at the *MF* and *INT* sites.

At *Soto del Frances*, the coarsest substrates were in the plots located in the upstream portion of the riparian area (*AGR*, *POP*, *MF*, and *CF* in Fig. 2). The levels of TOC and TN observed in the topsoil were highest in the *MF* and *INT* sites, and the *AGR* sites had the least organic substrate. As in the other areas, the *AGR* sites had the highest proportions of organic nitrogen. RIC was highest at the *MF* and *POP* sites, while those elevated AGR values were due to extreme values, as observed with the standard error (Tab. 2). As occurred in the other two study zones, ROC content depended on the quantity, rather than the quality, of the organic matter.

Sedimentation

Significantly more sediment was deposited at *Mejana de Pastriz* (MP), which is a point bar (Tab.3), than at *Rincón Falso* (RF) and *Margen Derecha* (MD), which are side-channels not permanently connected with the Ebro River (Fig. 4). Sediment texture and the concentrations of TOC and TN in deposited sediment were strongly correlated (Fig. 4), which influenced the quantity of the organic fraction

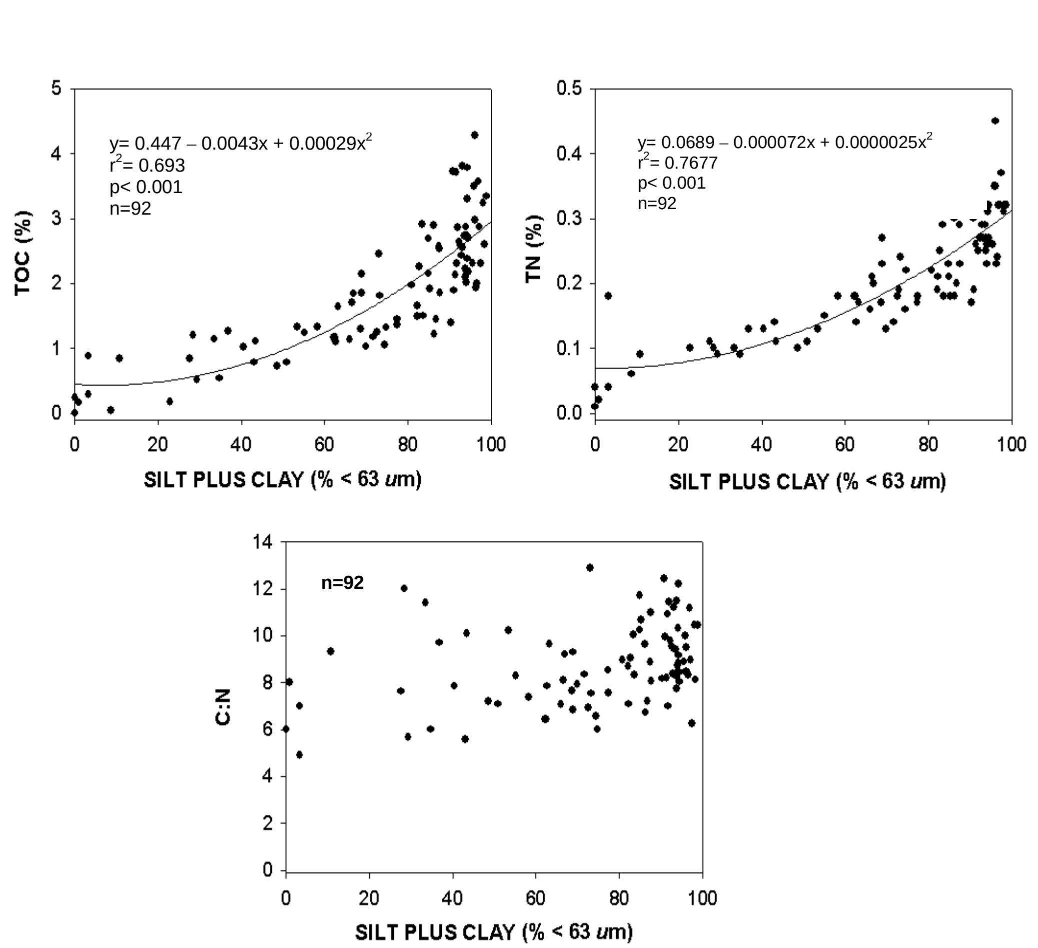

Figure 4. Scatter plots of sediment texture (proportion (%) of silt plus clay) and (a) TOC, (b) TN, and (c) C:N ratios in the sediments deposited during a flood on 1 January 2006 in the study area within the Middle Ebro River Floodplain, Spain. TOC= Total Organic Carbon; TN= Total Nitrogen; C:N= Carbon/Nitrogen ratio.

	Mean ± Standard Error		
	Soto Francés	*Margen Derecha*	*Mejana Pastriz*
Dry flood deposit (g m-2)	2330.70±771.08	2690.93±118.37	5987.41±776.7
Silt plus Clay (%<63 *u*m)	79.83±5.70	77.67±3.25	60.22±4.06
TOC (%)	2.75±0.18	2.14±0.12	1.27±0.09
TOC accretion rate (g C m-2)	63.24±4.48	58.97±6.52	70.59±7.95
TN (%)	0.29±0.02	0.24±0.01	0.16±0.01
TN accretion rate (g N m-2)	3.71±0.43	6.38±0.63	8.89±0.83
C:N	9.62±0.23	9.01±0.26	8.03±0.31

Table 3. Summary results for analyzed variables in deposited sediment during the studied event. Sampling Plots have been grouped by riparian area. TOC= Total Organic Carbon; TN= Total Nitrogen; C:N= TOC/TN

deposited because of the greater OM affinity for the finest particles (Tab. 3). However, sediment texture and the C:N ratio were not significantly correlated, although variability was higher at higher percentages of silt plus clay (Fig. 4).

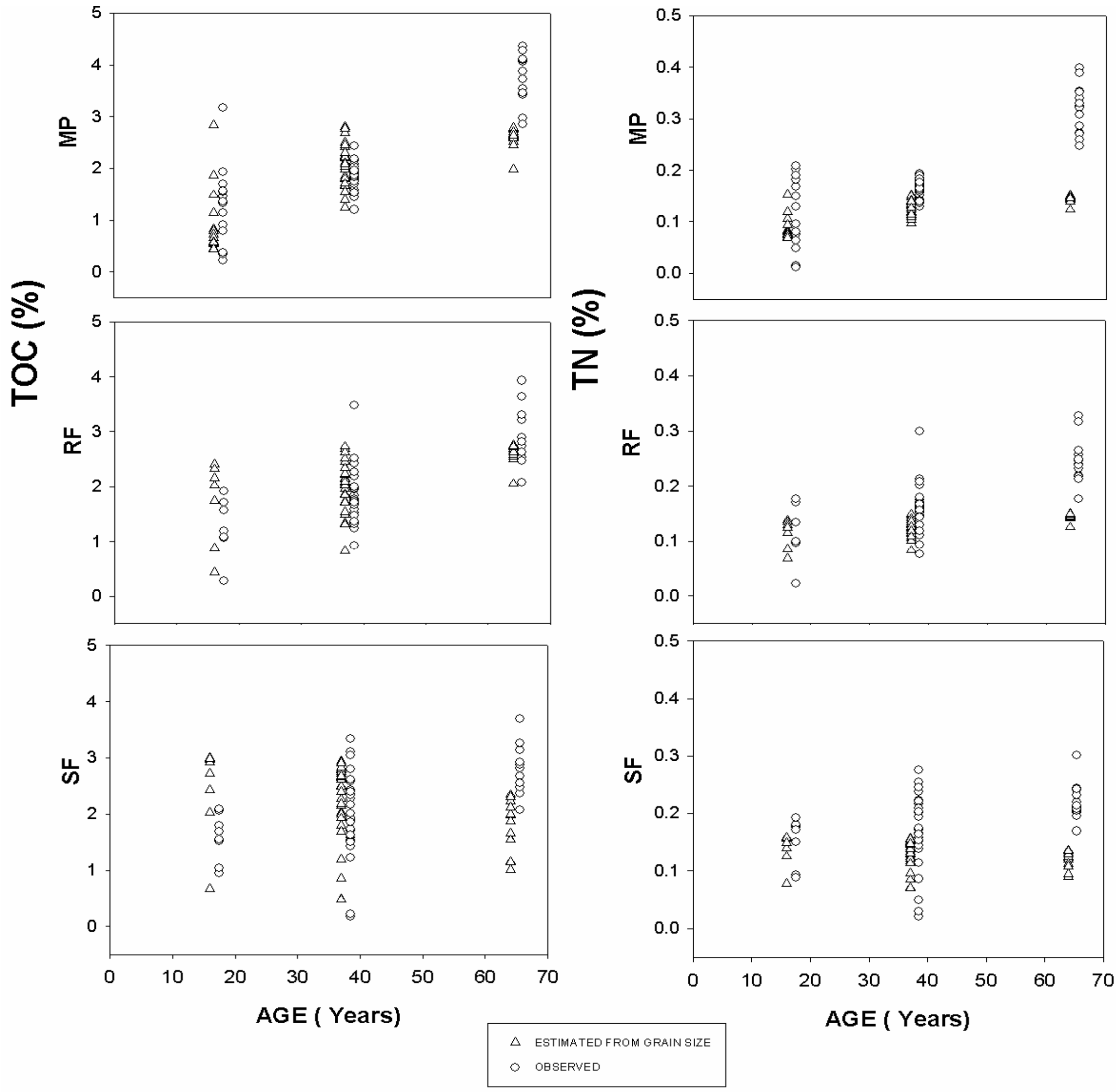

Figure 5. Proportions (%) of TOC and TN in the topsoil of natural habitats (see Fig. 4) at three areas within the Middle Ebro River Floodplain, Spain. The ages of the landforms at each of the sampling sites are based on an estimate of the date since the main channel migration. TOC= Total Organic Carbon; TN= Total Nitrogen; MP= Mejana Pastriz; RF= Rincon Falso; SF= Soto Francés

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Incorporation of organic matter produced in situ into the topsoil layer

The greatest differences between the observed and initial values of TOC and TN were in the older patches (>60 y) of the three examined areas (Fig. 5). Regardless the grain size (Tab. 2), these differences were similar among the mature forests *(MF)* in the three zones of the study, although, at Rincon Falso (RF), differences were slightly smaller. In the youngest patches, observed values were similar to initial despite

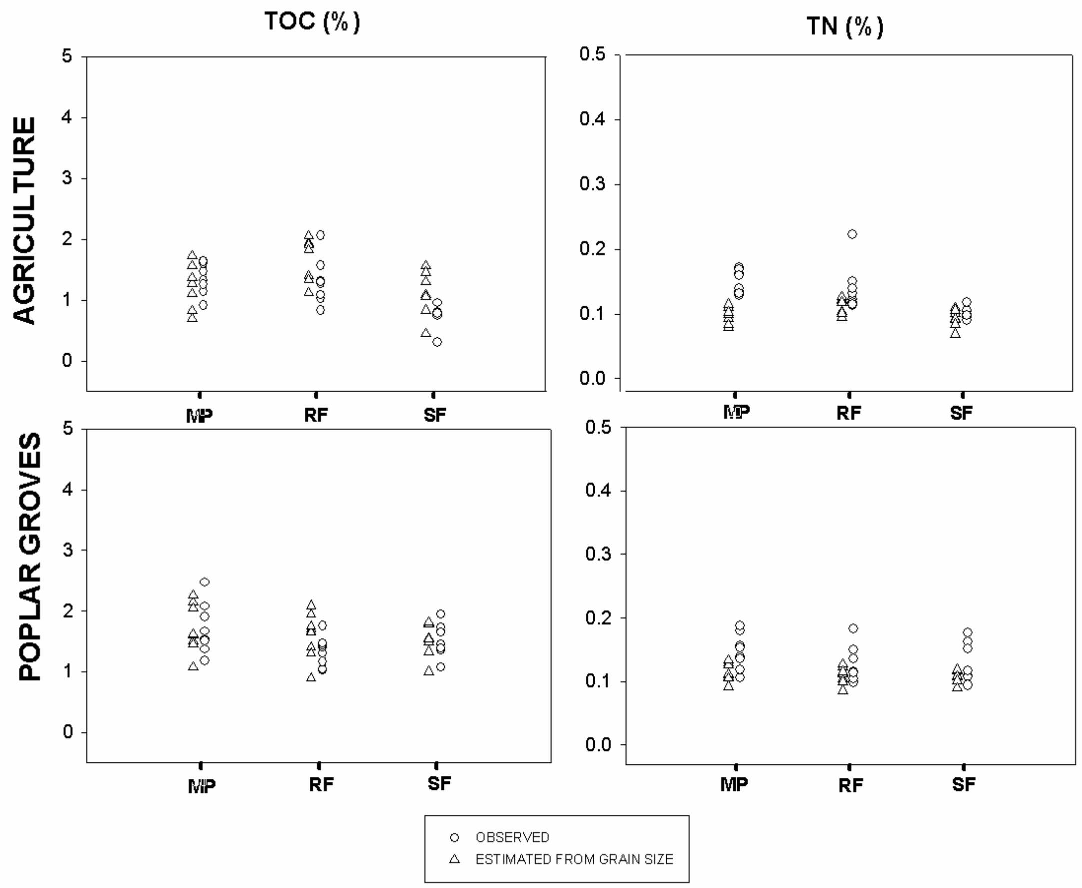

Figure 6 The proportions (%) of TOC and TN in the topsoil of human-affected plots at three locations within the study site on the Middle Ebro River Floodplain, Spain. TOC = Total Organic Carbon; TN = Total Nitrogen

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differences in the composition of grain sizes (Tab. 2). In the human-influenced plots (*AGR* and *POP* in Fig. 2), however, the differences among riparian areas were due to differences in sediment texture, only because TOC and TN values were similar for the topsoil concentrations and the predictions from deposited sediments (Fig. 6). The budgets of organic carbon and nitrogen in agricultural fields and poplar groves were similar to those in the youngest natural plots (Fig. 5 and Tab. 2).

Discussion

Soil Organic Matter (SOM) in natural land-cover categories

Our study in the Middle Ebro River Floodplain showed that the development of landforms, which influences the spatial differences in hydrological connectivity between the river and the floodplain, explained most of the variability in the amounts of Soil Organic Carbon and Nitrogen (SOC and SON) in the topsoil of natural habitats. The position of the plots in the ecotopes within the riparian succession (the "Category") had a significant effect on SOC and SON storage. In addition, the location within the reach on the Middle Ebro River (the "study zone") had also a significant effect, which reflected the importance of processes that occur over broader spatio-temporal scales, such as large floods or groundwater dynamics. Also the interaction of seral state and location had a significant effect on TOC and TON, which reflected the variability in the differences among the categories in the selected areas. In the initial stages of wetland formation, the specific hydro-morphological conditions determine the chemical and physical properties of the substrate, which leads to a specific biotic ecosystem that modifies the wetland hydroperiod iteratively, therefore conditioning the quantities of SOC and SON in the topsoil (Mitsch and Gosselink, 1993). In our study, however, variability in the biochemical complexity of the SON and SOC, as reflected by the C:N and RIC, might have been caused by factors other than those evaluated in our study. The SOC *passive* fractions suggest that the factors governing the variability in the quantity, rather than the quality, of SOC were more important in the formation of the recalcitrant pool at the study area.

The significant relationship of soil texture (silt plus clay) and TOC, TN and ROC reflects the importance of river inputs in the formation of the floodplain substrate. In recently deposited sediments, particle size, TOC, and TN were strongly positively correlated. Betchold and Naiman (2006) observed that TOC and TN storage in the riparian soils of a semi-arid savanna were strongly correlated with the concentrations of fine particles, which was due to the high affinity of organic matter for the finest fraction of the river seston during sedimentation (Walling *et al*, 1997; Steiger and Gurnell, 2003). Our results demonstrated that the incorporation of organic carbon and nitrogen produced *in situ* was evident in the topsoil of older patches that had low surface connectivity, regardless their grain sizes. In the light of RF results, further research is needed to confirm if this SOM pool remains constant for a long period. According to the results of the aerial pictures analysis, we concluded that the Mature Forest plot at this site is older than 68 years. It was not possible to compile information prior to 1927 for assigning an approximate date. In contrast, the proportions of TOC and TN in the topsoil of the connected categories did not differ markedly from TOC and TN considered as initial values. Thus, the current geomorphological dynamics still inhibit soil formation in the younger patches of the floodplain. The aggradation rate of floodplains is inversely correlated with soil development (Daniels, 2003). Our results confirm that the net accumulation of carbon and nitrogen in riparian forest soils can be a function of the time since patch formation or disturbance (Giese *et al*, 2000; Wigginton *et al*, 2000; Balian and Naiman, 2005).

At the study area in the Ebro River, the stocks of SOC and SON were highest in the topsoil of the substrates underlying the mature stages of vegetation, particularly, *Populus alba*. Typically, organic matter and nutrients accumulate in the older floodplain soils because of autogenic soil building (Chapin *et al*, 1994; Schwendenmann, 2000; Adair *et al*, 2004). In our study, SOC and SON levels were lowest in the intermediate-age patches, where the specific frequency, duration, and scouring effect of floods probably contributed to the small differences between Categories. Differences were the highest in the SF zone, where the substrates of highly connected patches were markedly coarser than those on intermediate connected substrates. Thus, it appears that the sedimentation rates and organic matter exports were greater in this plot, which was at the upstream portion of the SF, where the water scouring is strong. Despite differences in grain sizes, the lowest SOC and SON concentrations were in the most highly connected patches. The highest accretion rates occur in those types of patches (Asselman and Middelkoop, 1995), but so too are the rates of erosion and the export of inorganic sediments and organic matter during erosive floods. To understand the differences within natural patches, the effects of hydroperiod on the dynamics organic matter require further study (Megonigal *et al*, 1997; Burke *et al*, 1999; Robertson *et al*, 2001; Lockaby *et al*, 2005).

In our study, Plot Type and Study zone did not explain a significant amount of the variability in the C:N ratio (i.e., SOM quality) (Tab. 1). Furthermore, C:N ratios were not significantly correlated with the silt plus clay fraction in the topsoil or the sediment. However, differences in the relative importance of external inputs and autochthonous organic matter appear to have contributed to the variability in the C:N ratio, given that river inputs are expected to have lower C:N ratios than were the substrates of the natural land-cover categories. More labile compounds from aquatic communities (Hein *et al*, 2003; Knosche, 2006) might reduce the C:N ratio in the sinking material, but slightly higher C:N ratios in the highly connected plots might indicate rapid processing of those river inputs. In contrast, C:N ratios of up to 20 have occurred in the riparian communities of the study area (Gonzalez *et al*, 2006). Thus, differences in the exchange of organic matter between the main channel and the adjacent floodplain might explain some of the variability in the data, although large floods that affect the entire study area might reduce the differences between categories.

Our results indicate that the *passive* SOC pool (ROC in Tab. 2) relied on the quantity of organic carbon, which was around 50% (RIC in Tab. 2), rather than on its biochemical complexity. Typically, in temperate climates, mineral-related OM that persists for hundred of thousands of years constitutes one half or more of SOM (Trumbore, 1993). However, the variability in the RIC of natural categories was not consistent among the three study areas. At the study reach, the influence of litter fall was expected to be appreciably higher in older substrates than elsewhere because of the higher soil RIC. Indeed, those plots had relatively higher RIC values, but RIC values at connected patches were also high. Tan *et al* (2004) found that the proportion of NHC was higher in the forests than in the meadows of an experimental watershed. In contrast, Plante *et al* (2006) found no relationship between SOC and the proportion of NHC after natural habitats were converted to agriculture, which suggested that recalcitrant fractions played a minor role in protecting soil C. Rovira and Vallejo (2002) found that RIC decreased as decomposition proceeded using H_2SO_4 hydrolysis, although they did not observe any relationship using HCl hydrolysis. Consequently, more thorough analyses of the effect of organic matter processing on RIC and the quantification of other SOC pools are required before generalizations can be made about the relationship between SOC biochemical complexity and land-cover. Furthermore, in our study, the RIC and the silt plus clay fraction of topsoil samples were not significantly correlated. Also it has been observed elsewhere, SOC can exhibit different degrees of biogeochemical complexity irrespective of its potential physical protection (Paul *et al*, 2001; Plante *et al*, 2006; Rovira and Vallejo, 2007).

Riparian Soils as SOC and SON sinks

River floodplains can serve as sinks for organic matter produced in other parts of the watershed and *in situ* (Wallig *et al*, 1996; McCarty and Ritchie, 2002; Balian and Naiman, 2005). For instance, the high productivity of riparian ecosystems (Mitsch and Gosselink, 1993) can cause higher SOM accumulation rates than those that occur in upland forests. However, site conditions and the processes involved form a complex network of interactions that can cause high spatial and temporal variability in the accumulation of organic matter in different habitats of a floodplain. The amounts of TOC in the dry soil of mature habitats were similar to those recorded in the topsoil (depth = 18 cm) of broadleaved and coniferous forests (3.9%, n = 188) in Spain, which takes longer to develop, but were lower than proportions of TOC in the topsoil of shrublands (5.8%, n = 124) (Hontoria *et al.,* 1999). Furthermore, RIC values indicated that half of the SOC at the study area can be considered to be in Non Hydrolizable Carbon (NHC) pool. This organic matter fraction is thought to represent the older, recalcitrant C pool (Leavitt *et al*, 1996; Paul *et al*, 2001; Rovira and Vallejo, 2002; Tan *et al*, 2004; Paul *et al*, 2006) but, to quantify fully the stabilized SOC pool, physical protection must be evaluated further (Mikkuta *et al*, 2006).

In our study area, human occupation of the Ebro River region has led to smaller accumulations of SOC and SON during the intervals between floods. The amounts of SOC and SON in the topsoil of patches used for intensive agriculture and poplar production were similar to those in young natural patches, even though their age (>60 y old) since formation. Agricultural fields and poplar groves are usually protected against floods, so suspended sediments are not deposited in these patches, while primary production is not entirely logged on soils (Fig. 6).

A management framework for C and N accretion in river floodplains

The accumulation of organic matter, particularly carbon, is an interesting target which could be included in river-floodplain restoration plans. The potential of soil restoration has been estimated to sequester 0.003 Pg C per year, globally (IPCC 2000). To maximize the accumulations of SOC and SON in the Middle Ebro River Floodplain the conversion of agricultural fields and poplar groves into Natural land cover habitats is required, as well as the restoration of the natural dynamics at terrestrialized landforms (Fig. 7). Currently, the main channel is not migrating and, in the riparian zone, vertical accretion is the dominant process (Cabezas, in press), which has led to an inverse relationship between patch age and degree of lateral connectivity with a clear scarcity of young patches. As proposed by Steiger *et al*. (2001) for the Garonne River Floodplain in France, the riparian zone within the Ebro River Floodplain might be inundated by water during large floods, which changes the dynamics of sedimentation. Meanwhile, the distance between the water table and the top of the substrate increases leading to the senescence and dieback of riparian forests (e.g., James, 1996). In order to reverse this trends, the natural dynamics of the ecosystem has to be restored, so that the hydrological connectivity between the river and the floodplain is increased (Ward et al, 2001; Kondolf et al, 2006). Therefore, terrestrialized habitats could became part of the functional floodplain (Fig.7). It may be premature to conclude that an increase in the amount of mature forest will maximize the accretion of SOC and SON at the reach scale since no

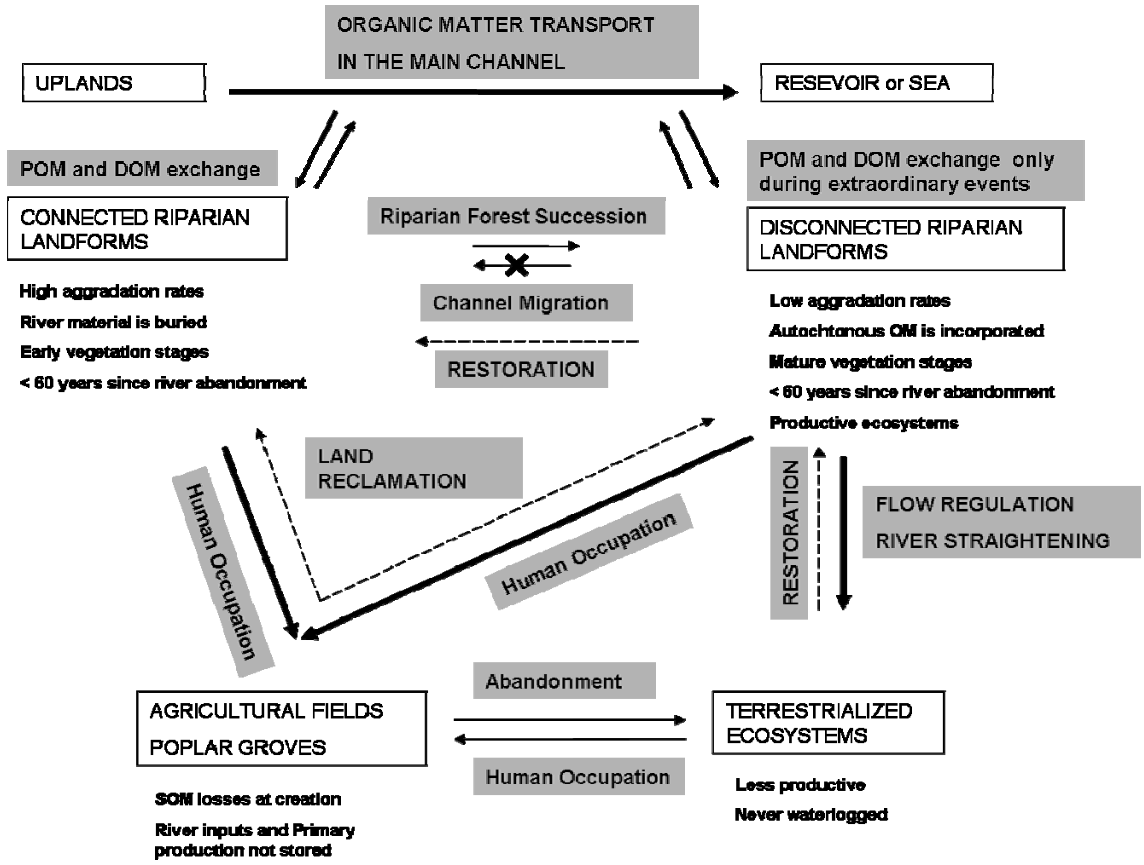

Figure 7 Conceptual scheme of the development of land-cover and management options within the study area on the Middle Ebro River Floodplain, Spain.

information about historical accretion rates is available. The way to enhance the accumulation of SOC and SON is therefore to increase the extent of the functional floodplain, it is the areas occupied by all natural categories. We hypothesize that this type of management might meet other restoration objectives, such as the maintenance of landscape heterogeneity and the enhancement of water quality.

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References

Amoros, C. and G. Bornette, 2002. Connectivity and biocomplexity in waterbodies of riverine floodplains. Freshwater Biology, 47: 761-776.

Amoros, C. and P.M. Wade, 1996. Ecological succesions. In: G.E. Petts and C. Amoros (Eds), Fluvial Hydrosystems, Chapman & Hall, London, pp. 211-241.

Adair, E.C., D. Binkley and D.C. Andersen, 2004. Patterns of nitrogen accumulation and cycling in riparian floodplain ecosystems along the Green and Yampa rivers. Oecologia, 139: 108-116.

Asselman, N.E.M. and H. Middelkoop, 1995. Floodplain Sedimentation - Quantities, Patterns and Processes. Earth Surf. Procc. Land., 20: 481-499.

Balian, E.V. and R.J. Naiman, 2005. Abundance and production of riparian trees in the lowland floodplain of the Queets River, Washington. Ecosystems, 8: 841-861.

Bechtold, J.S. and R.J. Naiman, 2006. Soil texture and nitrogen mineralization potential across a riparian toposequence in a semi-arid savanna. Soil Biol. Biochem., 38: 1325-1333.

Burke, M.K., B.G. Lockaby and W.H. Conner, 1999. Aboveground production and nutrient circulation along a flooding gradient in a South Carolina Coastal Plain forest. Canadian Journal of Forest Research.-Revue Canadienne De Recherche Forestiere, 29: 1402-1418.

Cabezas, A., E. Gonzalez, B. Gallardo, M. García, M. Gonzalez and F.A. Comín, in press. Effects of hydrological connectivity on sediment and understory structure in the middle Ebro river (NE Spain): Implications for restoration and management. Aquat Sci.

CEDEX, 1997. Estudio sobre el impacto potencial del Cambio Climático en los recursos hídricos y las demandas de agua de riego en determinadas regiones de España., Informe Técnico para el MIMAM, MADRID.

Chapin, F.S., L.R. Walker, C.L. Fastie and L.C. Sharman, 1994. Mechanisms of Primary Succession Following Deglaciation at Glacier Bay, Alaska. Ecol Monogr., 64: 149-175.

Comin, F.A., 1999. Management of the Ebro River Basin: Past, present and future. Water Sci. Technol., 40: 161-168.

Daniels, J.M., 2003. Floodplain aggradation and pedogenesis in a semiarid environment. Geomorphology., 56: 225-242.

Day, J.W., J.Y. Ko, J. Rybczyk, D. Sabins, R. Bean, G. Berthelot, C. Brantley, L. Cardoch, W. Conner, J.N. Day, A.J. Englande, S. Feagley, E. Hyfield, R. Lane, J. Lindsey, J. Mistich, E. Reyes and R. Twilley, 2004. The use of wetlands in the Mississippi Delta for wastewater assimilation: a review. Ocean & Coast. Manag., 47: 671-691.

Giese, L.A., W.M. Aust, C.C. Trettin and R.K. Kolka, 2000. Spatial and temporal patterns of carbon storage and species richness in three South Carolina coastal plain riparian forests. Ecol. Eng., 15: S157-S170.

Gonzalez, E., A. Cabezas, F.A. Comin, M. García and B. Gallardo, 2006. Structural characteristics of floodplain forest at patch and landscape scales as reference for Ecological Restoration. In: F. Bravo Oviedo (Ed), IUFRO DIV.4. International Meeting “Managing Forest Ecosystems: The Challenge of the Climatic Change”, Cuatro Vientos, Valladolid. p. 17.

Hein, T., C. Baranyi, G.J. Herndl, W. Wanek and F. Schiemer, 2003. Allochthonous and autochthonous particulate organic matter in floodplains of the River Danube: the importance of hydrological connectivity. Freshw Biol., 48: 220-232.

Hontoria, C., J.C. Rodriguez-Murillo and A. Saa, 1999. Relationships between soil organic carbon and site characteristics in peninsular Spain. Soil Sci. Soc Am J, 63: 614-621.

IPCC, 2000. Land Use, Land Use Change, and Forestry. Cambridge University Press, Cambridge. United Kingdom, 377 pp.

James, M., 1996. Le dépérerissenment des boisements riverains de la Garonne: évaluation à partir des données de structure forestière et de télétéction á haute résolution spatiale. PhD thesis, Univerité Paul Sabatier, Toulouse, p. 272.

Johnston, C.A., 1991. Sediment and nutrient retention by freshwater wetlands- Effects on surface-water quality. Crit. Rev. Environ. Control, 21: 491-565.

Junk, W.J., B. Bayley and R.E. Sparks, 1989. The flood pulse concept in river-floodplain systems. in D.P.Dodge (ed) Proc. Intern. Large River Symposium. Special Issue of J.Canad. Fish and Aquat. Sci., 106: 11-27.

Knosche, R., 2006. Organic sediment nutrient concentrations and their relationship with the hydrological connectivity of floodplain waters (River Havel, NE Germany). Hydrobiologia, 560: 63-76.

Kondolf, M.G., A.J. Boulton, S. O´daniel, G.C. Poole, F.J. Rahel, E.H. Stanley, E. Wohl, A. Bang, J. Carltroom, C. Cristoni, H. Huber, S. Koljonen, P. Louhi and K. Nakamura, 2006. Process-Based Ecological River Restoration: Visualizing Three-Dimensional Connetivity and Dynamic Vectors to Recover Lost Linkages. Ecol Soc, 11: 5.

Leavitt, S.W., R.F. Follett and E.A. Paul, 1996. Estimation of slow- and fast-cycling soil organic carbon pools from 6N HCl hydrolysis. Radiocarbon, 38: 231-239.

Lockaby, B.G., R. Governo, E. Schilling, G. Cavalcanti and C. Hartsfield, 2005. Effects of sedimentation on soil nutrient dynamics in riparian forests. J Environ Qual., 34: 390-396.

McCarty, G.W. and J.C. Ritchie, 2002. Impact of soil movement on carbon sequestration in agricultural ecosystems. Environ Pollut., 116: 423-430.

Megonigal, J.P., W.H. Conner, S. Kroeger and R.R. Sharitz, 1997. Aboveground production in Southeastern floodplain forests: A test of the subsidy-stress hypothesis. Ecology, 78: 370-384.

Mikutta, R., M. Kleber, M.S. Torn and R. Jahn, 2006. Stabilization of soil organic matter: Association with minerals or chemical recalcitrance? .Biogeochemistry, 77: 25-56.

Mitsch, W.J., J.W. Day, J.W. Gilliam, P.M. Groffman, D.L. Hey, G.W. Randall and N.M. Wang, 2001. Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River Basin: Strategies to counter a persistent ecological problem. Bioscience, 51: 373-388.

Mitsch, W.J., J.W. Day, L. Zhang and R.R. Lane, 2005a. Nitrate-nitrogen retention in wetlands in the Mississippi river basin. Ecol. Eng. 24: 267-278.

Mitsch, W.J. and J.G. Gosselink, 1993. Wetlands. Van Nostrand Reinhold, New York, 722 pp.

Mitsch, W.J., A.J. Horne and R.W. Nairn, 2000. Nitrogen and phosphorus retention in wetlands-ecological approaches to solving excess nutrient problems. Ecol. Eng. 14: 1-7.

Mitsch, W.J., L. Zhang, C.J. Anderson, A.E. Altor and M.E. Hernandez, 2005b. Creating riverine wetlands: Ecological succession, nutrient retention, and pulsing effects. Ecol. Eng., 25: 510-527.

Noe, G.B. and C.R. Hupp, 2005. Carbon, nitrogen, and phosphorus accumulation in floodplains of Atlantic Coastal Plain rivers, USA. Ecol Appl, 15: 1178-1190.

Ollero, A., 1992. Los meandros libres de el Ebro (Logroño-La Zaida). Geomorfología fluvial, ecogeografía y Riesgos. Tesis Doctoral. Geografía y Ordenación del Territorio, Zaragoza, Zaragoza, p. 1138.

Ollero, A., 1995. Dinámica reciente del cauce de el Ebro en la Reserva Natural de los Galachos. Cuaternario y Geomorfología, 9: 85-93.

Paul, E.A., H.P. Collins and S.W. Leavitt, 2001. Dynamics of resistant soil carbon of midwestern agricultural soils measured by naturally occurring C-14 abundance. Geoderma, 104: 239-256.

Paul, E.A., S.J. Morris, R.T. Conant and A.F. Plante, 2006. Does the acid hydrolysis-incubation method measure meaningful soil organic carbon pools? Soil Sci. Soc Am J., 70: 1023-1035.

Plante, A.F., R.T. Conant, E.A. Paul, K. Paustian and J.r. Six, 2006. Acid hydrolysis of easily dispersed and microaggregate-derived silt- and clay-sized fractions to isolate resistant soil organic matter. Eur J Soil Sci., 57: 456-467.

Robertson, A.I., P. Bacon and G. Heagney, 2001. The responses of floodplain primary production to flood frequency and timing. J Appl. Ecol., 38: 126-136.

Robertson, A.I., S.E. Bunn, P.I. Boon and K.F. Walker, 1999. Sources, sinks and transformations of organic carbon in Australian floodplain rivers. Mar. Freshw. Res., 50: 813-829.

Rovira, P. and V.R. Vallejo, 2000. Examination of thermal and acid hydrolysis procedures in characterization of soil organic matter. Commun. Soil Sci. Plant Anal., 31: 81-100.

Rovira, P. and V.R. Vallejo, 2002. Labile and recalcitrant pools of carbon and nitrogen in organic matter decomposing at different depths in soil: an acid hydrolysis approach. Geoderma, 107: 109-141.

Rovira, P. and V.R. Vallejo, 2007. Labile, recalcitrant, and inert organic matter in Mediterranean forest soils. Soil Biol. Biochem., 39: 202-215.

Rovira, P., C. Kurz-Besson, M.M. Couteaux and V.R. Vallejo, 2008. Changes in litter properties during decomposition: A study by differential thermogravimetry and scanning calorimetry. Soil Biology & Biochemistry, 40: 172-185.

Schwendenmann, L., 2000. Soil properties of boreal riparian plant communities in relation to natural succession and clear-cutting, Peace River lowlands, Wood Buffalo National Park, Canada. Water Air Soil Pollut., 122: 449-467.

Steiger, J. and A.M. Gurnell, 2003. Spatial hydrogeomorphological influences on sediment and nutrient deposition in riparian zones: observations from the Garonne River, France. Geomorphology, 49: 1-23.

Steiger, J., A.M. Gurnell, P. Ergenzinger and D. Snelder, 2001. Sedimentation in the riparian zone of an incising river. Earth Surf. Proc. Land., 26: 91-108.

Tan, Z.X., R. Lal, R.C. Izaurralde and W.M. Post, 2004. Biochemically protected soil organic carbon at the north appalachian experimental watershed. Soil. Sci., 169: 423-433.

Tockner, K., D. Pennetzdorfer, N. Reiner, F. Schiemer and J.V. Ward, 1999. Hydrological connectivity, and the exchange of organic matter and nutrients in a dynamic river-floodplain system (Danube, Austria). Freshw. Biol., 41: 521-535.

Tockner, K., F. Malard and J.V. Ward, 2000. An extension of the flood pulse concept. Hydrological Processes, 14: 2861-2883.

Trumbore, S.E., 1993. Comparison of Carbon Dynamics in Tropical and Temperate Soils Using Radiocarbon Measurements. Global Biogeochem. Cycles., 7: 275-290.

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

Walling, D.E. and Q. He, 1997. Investigating spatial patterns of overbank sedimentation on river floodplains. Water Air Soil Pollut., 99: 9-20.

Walling, D.E., Q. He and A.P. Nicholas, 1996. Floodplains as Suspended Sediment Sinks. In: G.M. Anderson, D.E. Walling and P.D. Bates (Eds), Floodplain Processes, Wiley, Chichester, pp. 399-440.

Ward, J.V. and J.A. Stanford, 1995. Ecological Connectivity in Alluvial River Ecosystems and Its Disruption by Flow Regulation. Reg. Rivers-Res. & Manag., 11: 105-119.

Ward, J.V., K. Tockner, U. Uehlinger and F. Malard, 2001. Understanding natural patterns and processes in river corridors as the basis for effective river restoration. Reg. Rivers-Res.h & Manag., 17: 311-324.

Wigginton, J.D., B.G. Lockaby and C.C. Trettin, 2000. Soil organic matter formation and sequestration across a forested floodplain chronosequence. Ecol. Eng., 15: S141-S155.

STUDY V

The effects of disrupting river-floodplain interactions on carbon and nitrogen accretion in riparian habitats

The effects of disrupting river-floodplain interactions on carbon and nitrogen accretion in riparian habitats

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Abstract

Sediment profiles from four riparian habitats in a reach of the middle Ebro River (NE Spain) were examined to study carbon (C) and nitrogen (N) accretion patterns during the last century and to assess the potential of floodplain substrates to act as organic matter (OM) sinks. Two oxbow lakes with different hydroperiods and two patches of riparian forest at different successional stages were selected to represent permanently and intermittently flooded habitats, respectively. Total organic carbon (TOC) and total nitrogen (TN) were analyzed to estimate OM quantities, and the non-hydrolyzable carbon approach was used to distinguish between labile and refractory soil OM pools. Cs-137 dating and aerial photographs were used to determine historical accretion rates. Our results indicate that human alterations at both basin and reach scales have severely modified C and N accretion patterns. Since 1963, the importance of allochthonous OM inputs to the study sites has diminished, indicating a reduction in the potential of the middle Ebro floodplains to act as C and N sinks. Sediment accretion rates were markedly higher in the 1927–1963 periods as were C and N accretion rates, despite the effect of post depositional OM processing. However, these differences did not result in quantitative differences in the refractory OM pool. Despite recent reductions in accretion rates, the middle Ebro floodplains retain a high potential to act as OM sinks, a fact that should be considered in future basin management plans. Suggestions to maximize C and N accretion through ecological restoration are provided.

Keywords: Cs-137, Ebro, floodplains, nitrogen, organic carbon, sedimentation.

Introduction

The major role of riverine floodplains as sinks, sources or transformers of organic matter (OM) has been highlighted by many researchers (Junk et al. 1989; Ward and Stanford 1995; Walling et al. 1996; Tockner et al. 1999; Robertson et al. 1999; Hein et al. 2003). However, studies that address the role of a floodplain as a permanent OM sink are scarce, despite the importance of organic carbon (C) and nitrogen (N) accumulation in these systems. In riparian ecosystems, nitrogen retention can improve the quality of surface water (Johnston 1991; Day et al. 2004; Verhoeven et al. 2006) and thereby prevent the eutrophication of downstream aquatic ecosystems. In terms of carbon retention, increasing the ability of ecosystems to act as carbon sinks has been suggested as a means of offsetting the greenhouse effect (IPCC 2000). There is therefore a need to quantify the long-term organic pools associated with riparian ecotopes. Soil organic matter (SOM) acquires special relevance in this context since it forms a larger and more lasting pool than OM accumulated as living biomass. SOM is also a complex pool, and includes plant, animal and microbes residues in all stages of decomposition, that are capable of combining with inorganic soil particles. Understanding this complexity is important in determining the influence of OM quality, for example, the degree to which SOM can be biodegraded (Rovira et al. 2008), on the potential of floodplains to act as OM sinks. Insights into these complex dynamics can be achieved using the acid hydrolysis approach, which is capable of distinguishing an older and more recalcitrant pool from a younger and more easily decomposable fraction (Paul et al. 2006). It is also important to know the effect of OM processing on the fate of the OM pool in floodplain substrates, both in terms of quantity and quality, in order to assess the role of floodplains as C and N sinks.

To evaluate C and N accretion in floodplain substrates, it is necessary to have a clear understanding of the effect of processes operating at different spatio-temporal scales. At the watershed scale, riverine floodplains can buffer rivers from the washload produced within the upstream parts of the catchment, although ongoing erosion and associated sediment remobilisation can impede the ability of riparian areas to act as permanent sinks (Steiger et al. 2005). The erosion and redistribution of OM within the watershed can have a major influence on OM dynamics, because riparian wetlands can support long-term budgets (McCarthy and Ritchie 2002). Interconnectivity between the different compartments of the basin is therefore a driving force for C and N transfer due to the importance of the fine-grained sediment dynamic. At the ecotope scale, OM produced by biotic assemblages is incorporated into surface soils in the intervals between floods. During floods, low-organic fluvial sediment may be deposited on riparian landforms, and OM stored in different compartments of the system (i.e., aboveground biomass, sediment and litter) is either exported or buried. As noted by Daniels (2003), the accretion rate is inversely related to soil development. The driving force that determines the relative importance of either autochthonous or allochthonous OM inputs is hydrological connectivity, which is a measure of the exchange of mass and energy between the river and its floodplain across different spatio-temporal scales (Junk et al. 1989; Tockner et al. 2000; Amoros and Bornette 2002). Under natural conditions, riparian succession towards disconnected ecotopes is counteracted by erosion and deposition during low-frequency floods. As a result, riparian landscapes contain units at every stage of succession (Amoros and Wade 1996; Ward et al. 2001). In the early stages, hydrological connectivity is high and substrates are therefore more inorganic due to high sedimentation rates; OM exports are also more frequent. In the later stages, ecotopes are less connected to the main channel and their substrates are more organic, due to incorporation into the soil pool of in situ-produced OM.

The effects of human alterations at basin and ecotope scales must also be considered, since they are likely to affect C and N accretion within floodplain systems. At the basin scale, changes in land-use and dam construction can alter the discharge of fine-grained sediment and dissolved nutrients, which influence C and N accumulation (Craft and Richardson 1998; Walling et al. 2003; Wang et al. 2004; Owens et al. 2005). Moreover, such alterations can modify the natural flow regime (Poff et al. 1997), producing dramatic effects on riparian succession. As a result, the diversity of accretion patterns in the floodplain and the exchange of OM between the river and the floodplain are also affected (Ward and Stanford 1995; Thoms 2003). At the ecotope scale, conversion of floodplain land to agricultural use implies construction of flood protection structures, which restrict OM exchanges to smaller areas over shorter time periods.

In this paper, we report a study of C and N accumulation patterns in the floodplain of a reach of the middle Ebro River in NE

Spain. During the 20th century, irrigation of lowland areas and abandonment of farmland in upland areas has resulted in dramatic changes in both the hydrology and the sediment load of the Ebro River (García-Ruiz et al. 1995; Ibañez et al. 1996; Beguería et al. 2003; Batalla et al. 2004; Beguería et al. 2006; Lopez-Moreno et al. 2006; Vericat and Batalla. 2006). Within the study reach, large flood control embankments have been built and reinforced after every large flood since the 1960s (Ollero, 1992). Against this background , we hypothesized that C and N accumulation patterns will have changed significantly within the study reach during the last century, affecting the potential of the middle Ebro floodplains to act as a sink for C and N. The objectives of the study were: a) to analyze the spatio-temporal heterogeneity of C and N accretion patterns; b) to assess the potential of the middle Ebro floodplains to act as C and N sinks; and c) to propose a valid management and rehabilitation framework that includes SOC and SON accumulation as restoration targets.

Materials and methods

Study Area

The study reach (Fig. 1) is located in the middle Ebro River, NE Spain. The Ebro River is the largest river in Spain (watershed area = 85,362 km^2, river length = 910 km, average annual discharge to the Mediterranean Sea = 14.442 hm^3/y) and remains geomorphologically active. The river meanders within the floodplain (sinuosity = 1.39, bank slope = 0.050%), which has an average width of 5 km (Ollero 1995). Within the study reach, the average monthly discharge is 230 m^3/s and the elevation ranges from 175 m a.s.l. in the river channel to 185 m a.s.l. at the base of the adjacent slopes. At the Zaragoza city gauging station (A011, 12 km upstream of the study reach), the potential storage capacity within the upstream catchment is 1.637.19 hm^3. This storage is mostly associated with two large dams constructed in 1945 and 1954, that are collectively capable of impounding approximately 50% of the mean annual runoff. At the study reach, the mean annual runoff and the bankfull discharge have declined approximately 30% since 1981. Agricultural fields and mature ecotopes have increasingly come to dominate (75% of floodplain area, unpublished data) over other patch types since 1957 (Fig. 1). Since 1981, there has been no lateral migration of the main channel (Fig. 1), further promoting the domination of mature ecotope stages (Cabezas et al. in press). The large flood control embankments constructed within the study reach since the 1960s (Ollero 1992) have further restricted the extent of overbank inundation.

Four riparian habitats with different geomorphological features were selected to represent the range of hydroperiod types at the study reach (Fig. 1). Two oxbow lakes with different hydroperiods were selected to represent permanently flooded areas (Fig. 1, top). OL2 and OL3 become superficially connected with the main channel when river discharge exceeds 600 m^3/s (0.15-y flood) and 1100 m^3/s (0.41-y flood), respectively. Two patches covered by young and mature stages of riparian forest were selected to represent frequently and rarely flooded areas, respectively (Fig. 1, bottom). BANK is a patch of riparian forest at an early successional stage and is covered mainly by grass and shrubs (*Tamaryx* sp.). It is inundated during almost every flood event (i.e., at >400 m^3/s). FOR is a patch of riparian forest currently dominated by mature *Populus alba* trees that is flooded at 1600 m^3/s (1.47-y flood).

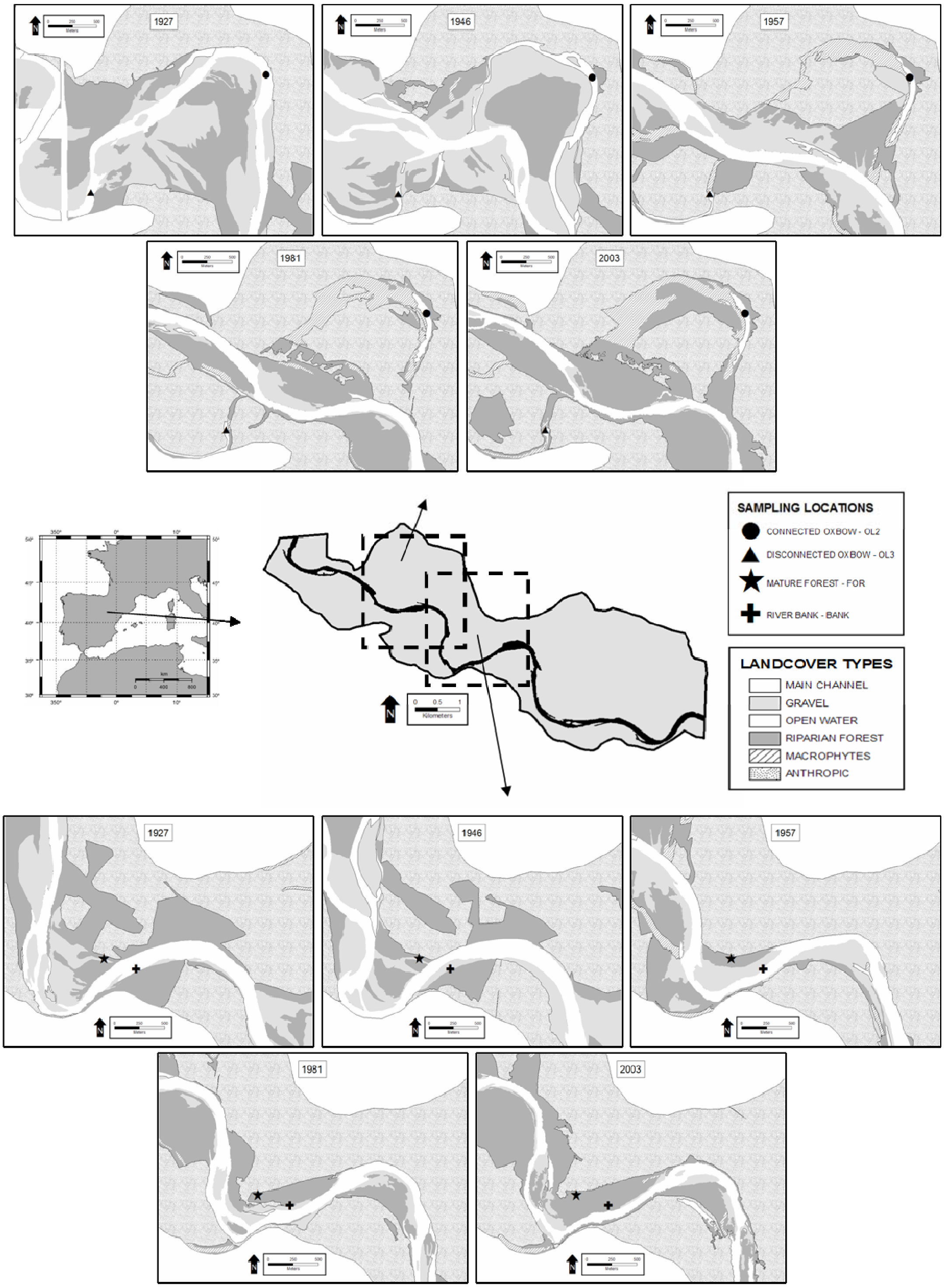

Figure 1. Location of the study sites in the examined reach, middle Ebro River (NE Spain). 1927, 1946, 1957, 1981 and 2003 land-cover maps are displayed to facilitate interpretation of the geomorphological evolution of selected ecotopes.

Sampling

Within each riparian habitat (n = 4), one sediment core was extracted at the lowest-lying elevation to allow comparisons between sampling locations. For the oxbow lakes, the sampling site was located at the point with the greatest waterdepth. Sediment cores (h = 240 cm) were collected in August 2006 (OL2) and August 2007 (OL3) from a boat using a Beeker undisturbed-sediment sampler

(Eijkelkamp®, ∅ = 57 mm). Sediment cores were carefully transported to the laboratory and extruded so as to avoid compaction, and sliced into 1-cm-thick sections. For the riparian forest habitats, the sampling location was selected after a field survey using a high-resolution GPS device (Top-Com®, ± 2 cm). Pits were dug in January 2007 (BANK) and August 2007 (FOR), reaching the coarse gravel layer at 80 cm and 280 cm in BANK and FOR, respectively. Sediment profiles composed entirely of fine sediment were extracted from the face of the trial pits using a three-sided frame (10 x 10 x 15 cm). The open end of the device was driven into the face and, after extraction, a blade was inserted to separate the sediment block into 1-cm increments. The sediment samples were stored in PVC bags and transported to the laboratory. All sediment samples were air-dried and weighed; those from BANK and FOR were screened through a 2-mm sieve.

Analyses

Historical accretion rates were estimated by evaluating the depth distribution of fallout Cs-137 in the sediment profiles. Significant fallout was first detected between 1953 and 1955 (Campbell et al. 1988). Fallout rates peaked in 1963–1964 and subsequently declined as a result of the 1963 Nuclear Test Ban Treaty (Walling and Bradley 1990). Global radioactive fallout rates have subsequently decreased steadily, except for minor increases in 1971 and 1974 associated with aboveground nuclear testing by non-treaty countries (Ritchie and McHenry 1990). The activity of Cs-137 in the sectioned cores was measured by gamma-ray spectrometry using a low-background HPGe detector. Count times were sufficient to provide measurements with a precision of ± ca 10% at the 95% confidence level. Cs-137 was measured down the profile until fallout could no longer be detected.

To investigate the spatial and temporal heterogeneity of accretion patterns, the depth distribution of C and N was also examined in the sediment cores. For each profile (n = 4), analyses were conducted every 2 cm (OL2, n = 120; OL3, n = 120; BANK, n = 40; FOR, n = 140). Grain-size analysis was performed on ungrounded samples using a laser-diffraction instrument (Coulter LS 230, Beckman Coulter®) after removing the organic matter with H_2O_2 and dispersing with sodium peroxide. Samples were then ground with a mortar and pestle for the remaining analyses. Total carbon (TC) was measured using an elemental analyser (Leco SC-144DR®), which was also used to determine Total Inorganic Carbon (TIC) in an aliquot heated at 550ºC for 4.5 h to eliminate organic matter. Total Organic Carbon (TOC) was calculated as the difference between TC and TIC. The magnitude of the recalcitrant OM pool was inferred from the biochemically protected TOC, using a single-step acid hydrolysis procedure (Rovira and Vallejo 2000). The results were used to calculate the Refractory Index for Carbon (RIC), which is expressed as the proportion (%) of non-hydrolyzed organic carbon (NHC). Total nitrogen (TN) was also measured using elemental analysis (Elementar Variomax CN®). C:N ratios were calculated by dividing TOC by TN to obtain a crude index of the organic matter quality. Estimates of organic carbon and nitrogen stocks in the profiles were made by multiplying TOC and TN by bulk density (BD), whereas estimates of Recalcitrant Organic Carbon (ROC) were calculated by multiplying RIC by organic carbon quantities.

Sediment, TOC, ROC and TN accretion rates were estimated using the Cs-137 dating, information from aerial photographs (Fig. 1), and the TOC, TN and ROC content of the sediment samples. Recent sedimentation rates (1963–present) were calculated as follows: First, using the well-defined Cs-137 fallout peaks, accretion rates were estimated for OL2, OL3 and FOR. For BANK, the gravel bed layer was estimated to have been deposited in approximately 1981, based on information on the geomorphological evolution of the area (Fig. 1). These values were used to calculate the minimum accretion rate since 1981. Second, early sedimentation rates (1927–1963) were estimated. At FOR, the fine sediment immediately overlying the gravel bed (h = 280) was deposited before 1927, since FOR was not part of the main channel at that period. This was used to estimate the maximum accretion rate for the 1927–1963 period, also taking into consideration the depth of the Cs-137 peak. Although the basal gravel layer was not reached by the cores collected from OL2 and OL3, the depth of the fine sediment deposit was established to be at least 4-m, by probing with a thin metal rod (∅ = 12 mm). This value and the Cs-137 peak were used to estimate minimum sediment accretion rates since both oxbows were known to be part of the main channel in 1927. Note that C and N accretion rates will be underestimated at OL2 and OL3, since only the upper 240 cm (where TOC,TN and ROC had been estimated) were included in the calculations.

		OL2	**OL3**	**FOR**	**BANK**
H (cm)	Cs-137 peak	91	69	25	-
	Cs-137 detection	139	91	85	-
	Profile	>400	>400	280	80
Accretion rate (cm/y)	1981-2007	-	-	-	>3.08
	1963-2007	2.07	1.57	0.57	-
	1927-1963	> 8.58	> 9.19	<7.08	-
TOC accretion rate (g C/m^2 y)	1981-2007	-	-	-	>300.1
	1963-2007	274.6	159.2	144.0	-
	1927-1963	>296.8	>379.2	<459.4	-
TN accretion rate (g N/m^2 y)	1981-2007	-	-	-	>31.6
	1963-2007	31.8	18.2	14.6	-
	1927-1963	>48.7	>44.3	<61.4	-
ROC accretion rate (g C/m^2y)	1981-2007	-	-	-	>160.6
	1963-2007	140.1	68.9	79.7	-
	1927-1963	>157.1	>152.3	<278.1	-

Table 1. Recent and early sediment and organic matter accumulation rates for the examined riparian habitats. (Total organic carbon, TOC; total nitrogen, TN; and recalcitrant organic carbon, ROC.)

To estimate the incorporation of in situ-produced organic matter into the sediment, an Organic Matter Incorporation index (OMI) was calculated as follows: OMI = (TOC x TN) / Fines (% < 63 μm). The organic load of the sediment, calculated as the product of TOC and TN (Rostan et al. 1987), was corrected for the magnitude of the fine sediment (< 63 *u*m) fraction due to the higher organic matter accretion associated with this fraction when river sediment settles (Asselman and Midlekoop 1995; Walling and He 1997; Steiger and Gurnell 2003). To test the theoretical influence of grain size on organic matter accretion, the relationship between TOC, TN, RIC and CN values and the magnitude of the fine fraction was examined. To analyze the effect of the dominant OM inputs on OM quality, the relationships between RIC and CN and OMI were also examined.

Samples were differentiated by habitat and deposition periods for all comparisons. The depth distributions of fines, TOC, TN, BD, RIC, CN and OMI depth profiles were all established. Finally, periods with different C and N accretion patterns were identified for the individual habitats using the previous results, with the initial appearance of Cs-137 activity being used as an approximate indicator of the early 1950s (Ritchie and McHenry 1990).

Results

Accretion Rates

The rate of fine sediment accretion decreased markedly during the last century at the study sites (Table 1). The 1963 Cs-137 peak was identified in the OL2, OL3 and FOR profiles, but not at BANK, where Cs-137 activities were the lowest (Fig.2). Accretion rates since 1963

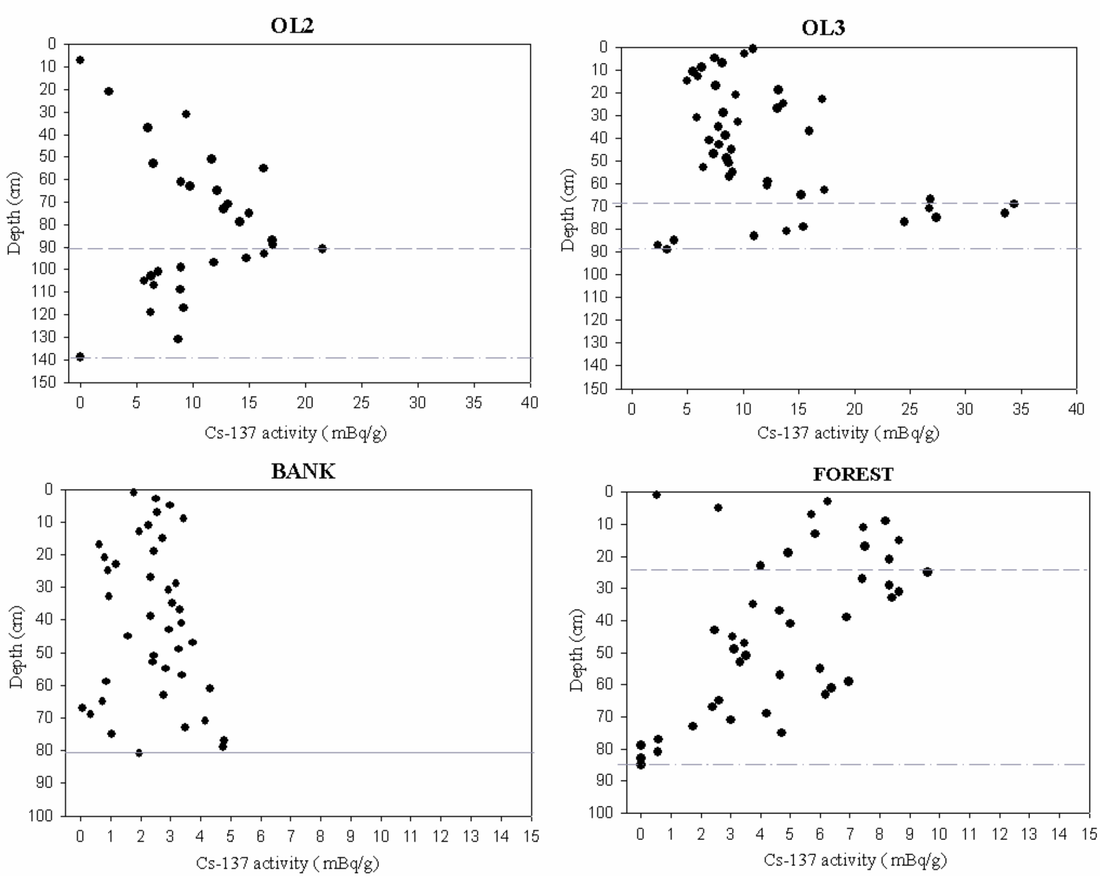

Figure 2. Cs-137 fallout (mBq/g) variation with depth (cm). Dashed lines indicate Cs-137 fallout peak (1963). Dashed-dotted lines indicate the extinction of Cs-137 fallout (early 1950s). Solid line at BANK indicates the depth of the gravel bed

were found to be highest at BANK and lowest at FOR, whereas OL2 and OL3 showed intermediate values, with the former being slightly higher. Prior to 1963, the depth of sediment deposited in the oxbow lakes was estimated to be thicker than at FOR.

Similarly, C and N accretion rates have significantly reduced since 1963 in all habitats (Table 1). After 1963, TOC accumulation was highest at BANK, followed by OL2, OL3 and FOR. Similar trends were found for TN and ROC. Although BANK and OL2 showed very similar TN accretion rates, biochemically protected carbon accretion was slightly higher at FOR than in OL3. Prior to 1963, accretion rates in the oxbow lakes were relatively high compared with those at FOR. Note that TOC, TN and ROC accretion rates for OL2 and OL3 are underestimated for the period before 1963 because only the upper 240 cm was used for the calculations.

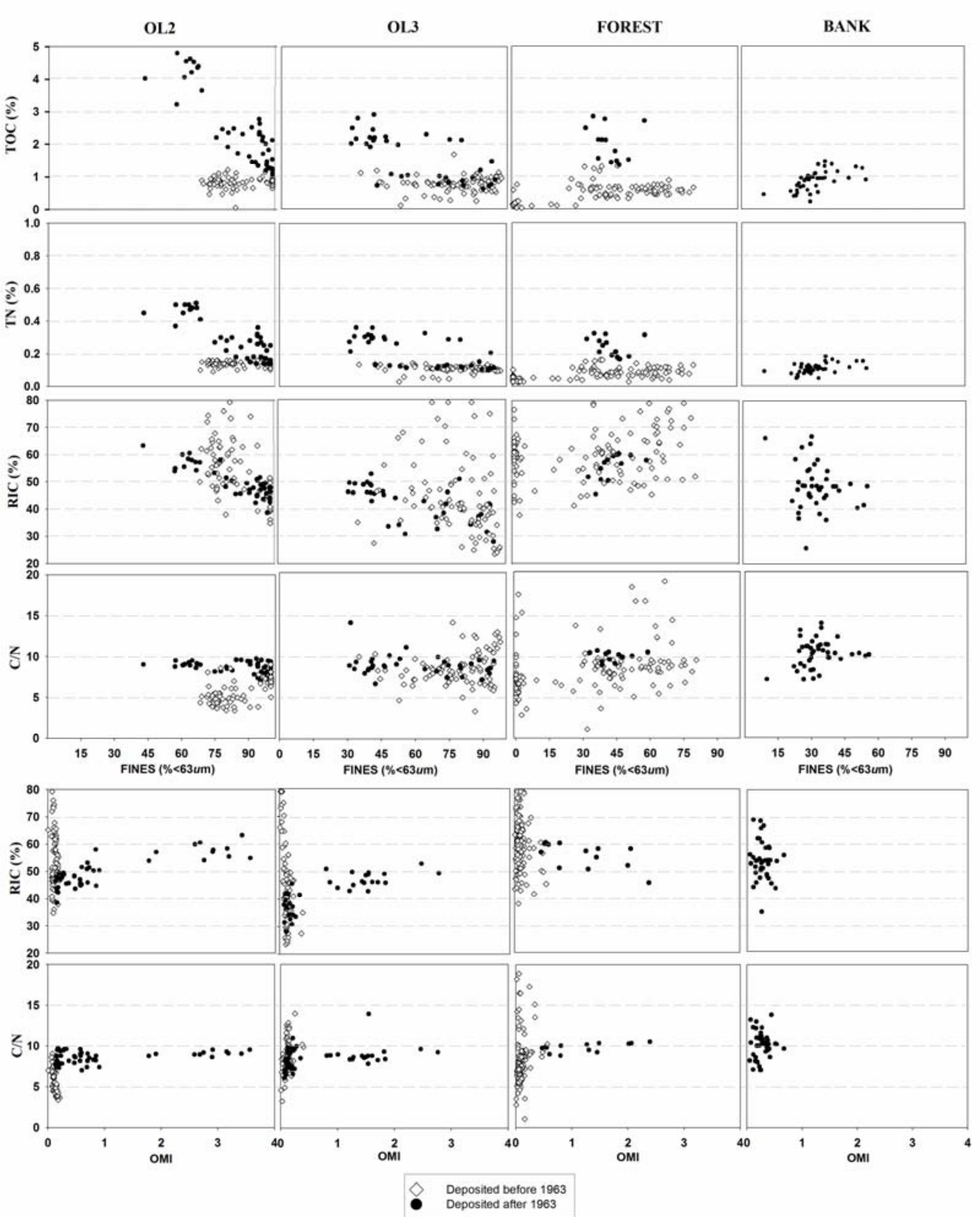

Figure 3. Relationship between grain size (% < 63 μm) and variables related to organic matter quantity and quality (top). Relationship between incorporation of autochthonous organic matter (OMI) to the substrate (see methods for details on OMI calculation) and variables related to organic matter quality (bottom). Samples are classified by deposition period.

Profile characterization

For most examined series, OM quantity and quality were unrelated to the sediment grain size distribution (Fig. 3, top). TOC and TN were significantly related to the proportion of fines only at BANK. In contrast, TOC and TN appeared to be positively correlated with the magnitude of the coarse fraction for recent sediment at OL2. With respect to RIC and C/N, a substantial amount of variability was found for sediment deposited prior to 1963 at OL2, OL3 and FOR, as well as for those deposited more recently at BANK. Focusing on the importance of different OM inputs, the incorporation of organic matter into the substrate was higher in recently deposited sediment (Fig. 3, bottom). At BANK, where the sediment was estimated to have been deposited after 1981, OMI values were estimated to be lower compared with recent sediment at the other study sites. However, no clear relationships were found between OMI and OM quality, with the exception of the recently deposited sediment from the oxbow lakes, where more highly organic substrates were associated with higher carbon biochemical complexity.

In terms of depth profiles, the four study sites sites showed different trends for variables related to OM quantity (Fig. 4; TOC, TN and BD), OM quality (Fig. 5; RIC and C/N) and autochthonous OM incorporation (Fig. 6; OMI). Oxbow lakes exhibited the finest substrates, although grain size fluctuations down the profile were more frequent and marked at OL3 (Fig. 4). At OL2, there was an abrupt shift towards a fines-dominated substrate around the 1950s, extending from a depth of 80 cm to 140 cm. After a second deposition of fine material (60–30 cm depth), the sand content markedly increased. Both fine sediment deposits were associated with relatively more dense material. Values for OMI, TOC and TN started to increase after 1963, after being nearly constant prior to that date (Figs. 4 and 6). Since 1963, the occurrence of fine-grained sediment peaks promoted relative decreases in OMI, TOC and TN. The last fines-dominated deposit (60–30 cm) was associated with a marked decrease in OMI, TOC and TN, which reached their highest values for all samples from the upper 30 cm of OL2. This event also induced an increase in RIC values, which had been relatively low compared with levels before the first massive deposition of fines (80–140 cm) in the early 1950s.

At the disconnected oxbow lake (OL3), OMI, TOC and TN increased at a depth of 50 cm (Figs. 4 and 6). At 30 cm, an episode of fines deposition promoted a decrease in the value of these variables, which were at their maximum above this level. Below 50 cm, OMI and TN values remained nearly constant, whereas TOC exhibited wider fluctuations that coincided with peaks in the incidence of fines. With respect to OM quality, RIC and C:N median values remained constant down the profile, although variability increased at lower levels. Below a depth of 70 cm, peaks associated with the RIC and C/N oscillations appeared to be inversely and directly related, respectively, with oscillations in fines.

At the intermittently flooded habitats (FOR and BANK), the substrate was coarser compared with that found in the oxbow lakes (Fig. 4), especially in the lowest part of the FOR profile, where sands completely dominated (180–208 cm depth). Within the upper part of the FOR profile (40-cm depth), the sedimentation of finer material caused a trend toward increased OMI (Fig. 6). In this region (40–0 cm), TOC and TN content increased and BD decreased. Unlike OMI fluctuations, the values of these variables were unrelated to oscillations in fines. Below 40 cm, TOC and TN values fluctuated within similar ranges to those

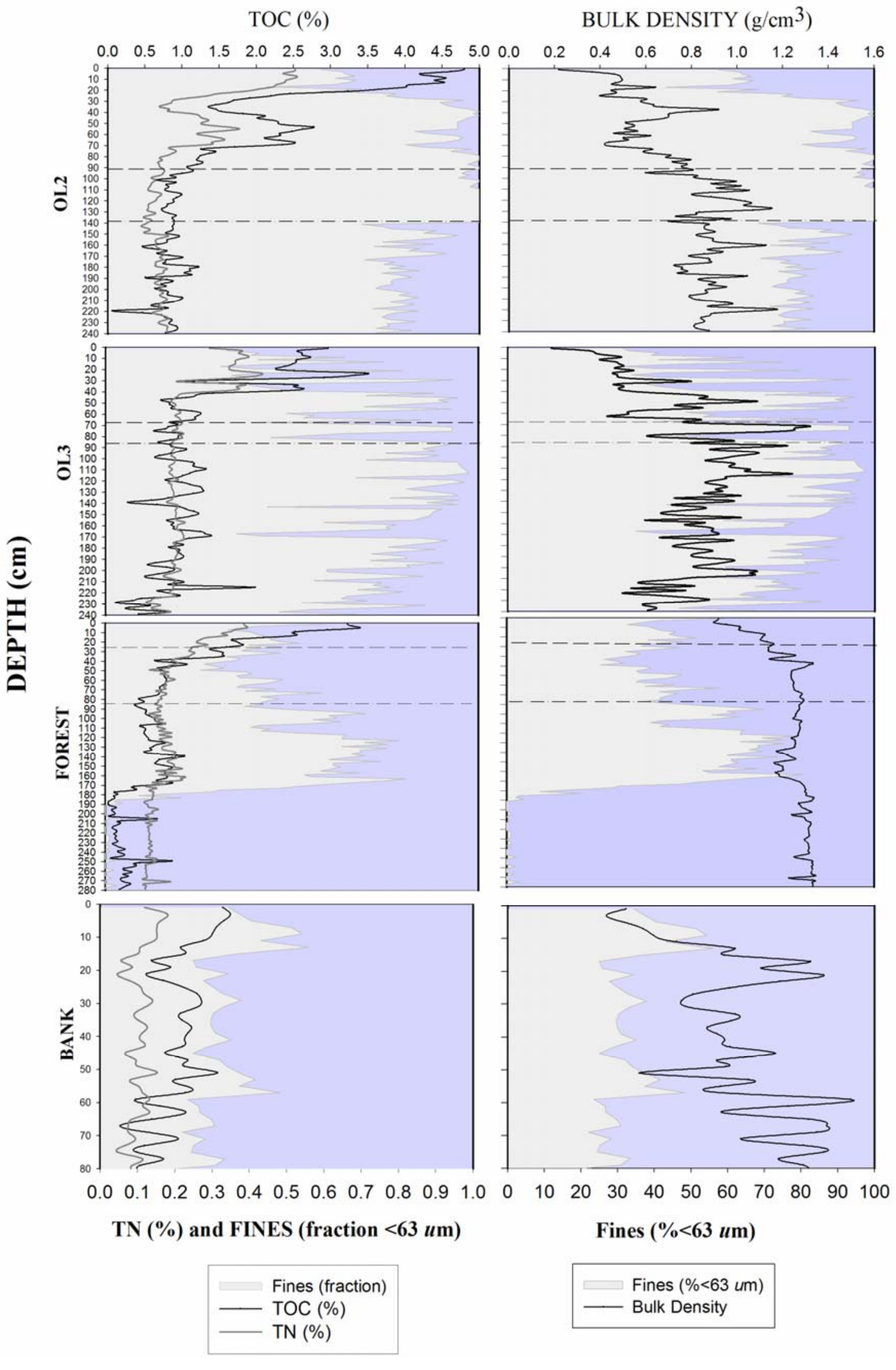

Figure 4. Organic carbon (%), total nitrogen (%) and bulk density (g/cm3) variation with depth (cm). Fines content (fraction < 63 µm) is also displayed as a vertical area plot. Dashed lines indicate Cs-137 fallout peak (1963). Dashed-dotted lines indicate the extinction of Cs-137 fallout (early 1950s).

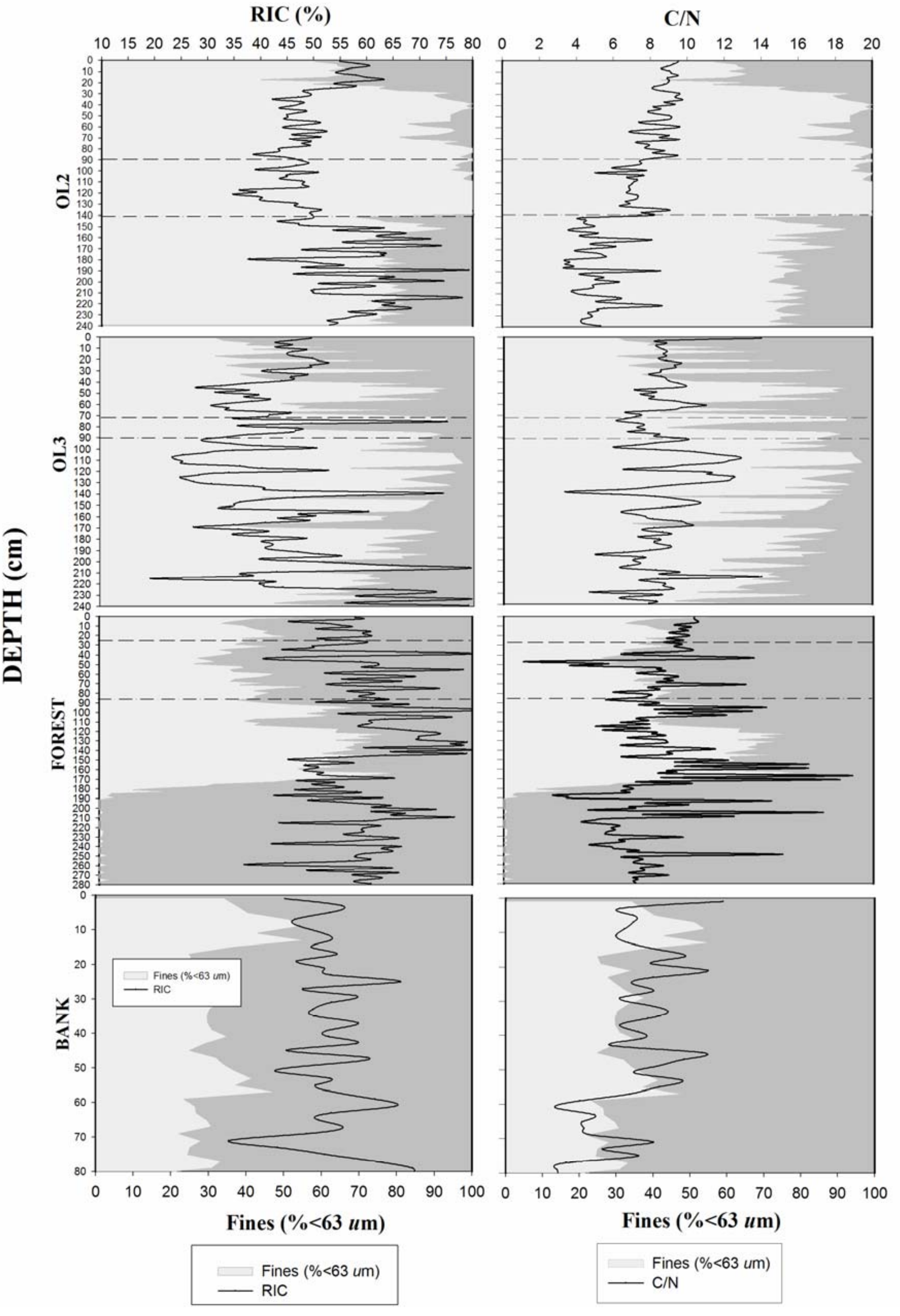

Figure 5. Recalcitrant carbon index (%) and C:N ratio variation with depth (cm). Fines content (% < 63 μm) is also displayed as a vertical area plot. Dashed lines indicate Cs-137 fallout peak (1963). Dashed-dotted lines indicate the extinction of Cs-137 fallout (early 1950s).

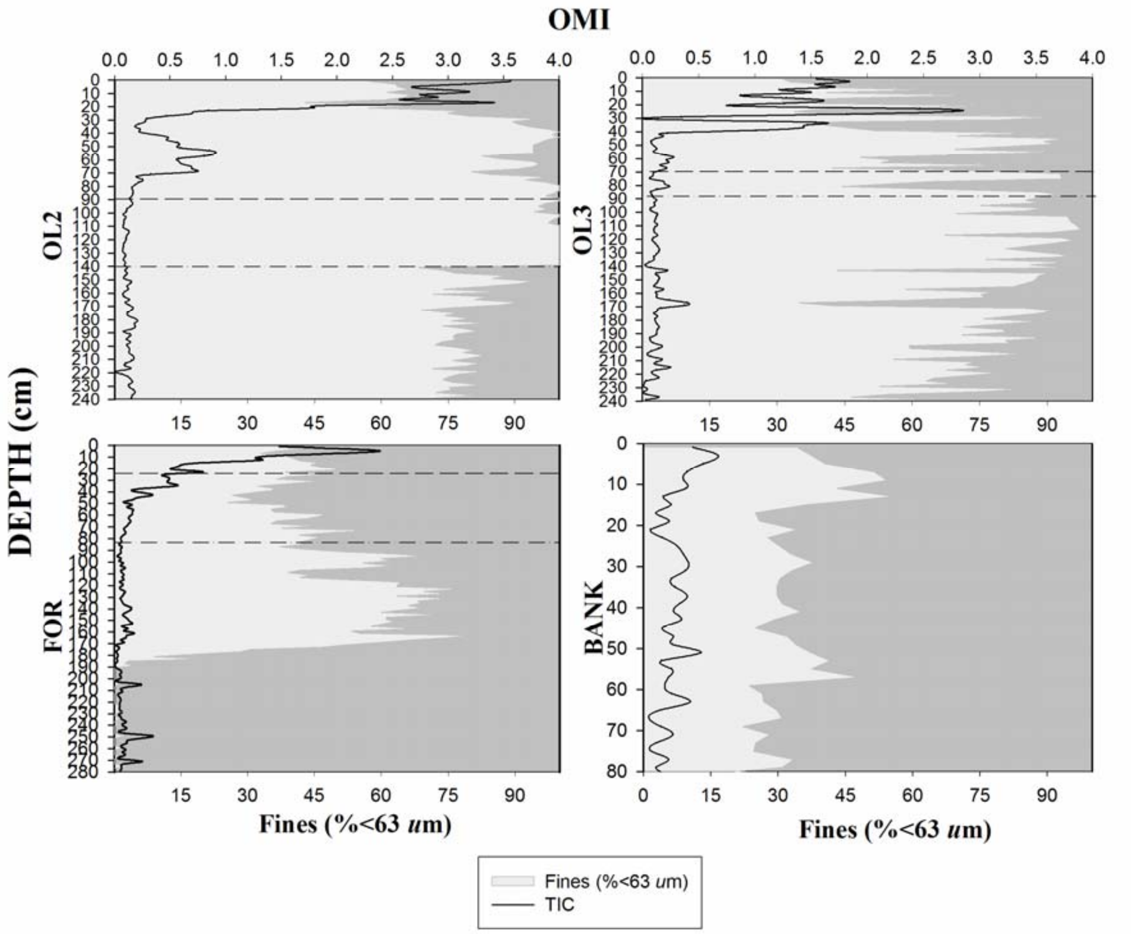

Figure 6. Variation in incorporation of autochtonous organic matter (OMI, see methods for details on calculation) with depth (cm). Fines content (% < 63 μm) is also displayed as a vertical area plot. Dashed lines indicate Cs-137 fallout peak (1963). Dashed-dotted lines indicate the extinction of Cs-137 fallout (early 1950s)

found in the lowest part of the oxbow profiles, with the exception of a sharp decrease in TOC in the deepest, sandy layer. The values of OM quality indicators, RIC and CN (Fig. 5), were highly variable, especially below a depth of 40 cm, generating highly fluctuating profiles. Finally, in the recently deposited BANK substrate, the profiles showed no evidence of abrupt changes. OMI values did not appear to be affected by the portion of fine sediment to the same extent as TOC and TN, which appeared to be directly related to the fines content. BD increased in the presence of sand.

Spatio-temporal heterogeneity

To characterize heterogeneity within the study reach, each profile was divided into sub-sections containing different C and N accretion patterns (Table 2). Since 1963, a thicker layer of finer sediment has been deposited within the oxbow lakes compared with FOR. However, TOC and TN concentrations were similar for all study sites, resulting in higher OMI values at FOR, which also exhibited a greater increase in the amount of carbon due to its heavier substrates. At BANK, a deep, bed of dense coarse inorganic material has been deposited since 1981, although OC and N concentrations were more similar to those that had accumulated earlier at the other study sites. In terms of OM quality, OL3 exhibited slightly lower levels of recalcitrant carbon.

For the period extending from the early 1950s to 1963, the thickness of the layer of deposited sediment layer was lowest at OL3. At OL2, the material deposited by the river was completely dominated by fines. TOC and TN concentrations were substantially decreased at all study sites, as were OMI values, resulting in a decrease in the differences in TOC and TN stocks between FOR and the oxbow lakes. BD increased at the three sites, although the FOR site was again associated with a more dense substrate. In terms of OM quality, RIC values were higher at FOR whereas RIC variability was two-fold higher at FOR and OL3 compared with the previous period.

MEAN ± STANDARD DEVIATION						
	OL2			OL3		
PHASE	**1 (n = 46)**	**2 (n = 24)**	**3 (n = 50)**	**1 (n=35)**	**2 (n = 11)**	**3 (n = 74)**
Depth (cm)	0-91	91-139	139-241	1-69	69-91	91-241
Period	After 1963	1950-1963	Before 1963	After 1963	1950-1963	Before 1963
Bulk Density (g/cm^3)	0.58±0.14	0.91±0.13	0.88±0.09	0.59±0.19	1.01±0.24	0.85±0.15
TIC (%)	4.38±0.49	3.73±0.37	3.95±0.18	3.69±0.27	3.72±0.26	3.73±0.42
TOC (%)	2.41±1.14	0.87±0.10	0.81±0.19	1.91±0.85	0.88±0.13	0.93±0.30
TN (%)	0.28±0.12	0.12±0.01	0.14±0.02	0.21±0.09	0.12±0.01	0.11±0.02
Clay (%<4*u*m)	30.07±13.66	43.30±6.99	14.06±3.90	13.12±6.22	16.36±6.68	15.26±6.37
Silt (63>%>4 *u*m)	55.41±9.56	55.84±6.99	64.70±4.49	46.68±16.73	62.03±12.47	62.81±9.78
FINE (%<62 *u*m)	85.48±15.42	99.14±1.63	78.76±5.81	59.79±21.59	78.38±17.80	78.07±14.38
RIC (%)	49.65±5.64	44.86±4.72	58.32±9.05	42.23±6.47	43.57±12.10	43.93±14.37
C/N	8.64±0.72	7.12±0.78	5.03±1.17	8.85±1.24	7.52±1.02	8.64±2.03
TOC (mg C/cm^3)	12.84±4.51	7.76±1.03	6.96±1.20	10.01±2.91	8.71±2.03	7.93±2.84
TN (mg N/cm^3)	1.49±0.49	1.10±0.14	1.23±0.17	1.15±0.33	1.17±0.28	0.91±0.24
ROC (mg C/cm^3)	6.54±3.02	3.46±0.47	3.99±0.65	4.34±1.75	3.74±1.15	3.15±0.79
OMI	1.00±1.08	0.10±0.02	0.13±0.04	0.87±0.78	0.13±0.06	0.14±0.07

	FOR				BANK
PHASE	**1 (n = 13)**	**2 (n = 30)**	**3 (n = 49)**	**4 (n = 48)**	**1 (n = 40)**
Depth (cm)	1-25	25-85	85-181	181-280	1-80
Period	After 1963	1950-1963	Before 1950	Around 30´s	After 1981
Bulk Density (g/cm^3)	1.03±0.09	1.24±0.05	1.25±0.04	1.31±0.02	1.00±0.27
TIC (%)	4.09±0.07	3.79±0.19	4.03±0.48	2.44±0.21	3.97±0.26
TOC (%)	2.42±0.66	0.87±0.33	0.61±0.21	0.22±0.16	1.05±0.36
TN (%)	0.24±0.06	0.10±0.03	0.08±0.03	0.04±0.01	0.11±0.03
Clay (%<4*u*m)	6.43±1.33	4.51±1.36	5.52±2.65	0.19±0.29	4.85±1.85
Silt (63>%>4 *u*m)	36.99±5.97	34.57±6.34	51.08±13.94	0.45±0.85	28.14±7.20
FINE (%<62 *u*m)	43.42±6.89	39.08±7.08	56.61±15.93	0.64±1.09	32.99±8.85
RIC (%)	55.73±4.61	59.72±8.96	62.10±10.38	58.97±7.53	53.61±6.97
C/N	9.80±0.55	8.27±2.34	9.19±3.14	7.30±2.72	10.15±1.66
TOC (mg C/cm^3)	24.37±4.59	10.66±3.56	7.59±2.40	2.82±1.96	9.54±1.57
TN (mg N/cm^3)	2.48±0.39	1.20±0.36	1.02±0.33	0.51±0.17	1.00±0.23
ROC (mg C/cm^3)	13.49±2.32	6.24±1.81	4.76±1.77	1.67±1.20	5.09±1.00
OMI	1.21±0.65	0.20±0.15	0.08±0.04	0.08±0.07	0.27±0.13

Table 2. Mean and standard deviation for different soil variables at different riparian ecotopes. Samples have been grouped into periods with different accretion patterns.

Prior to the 1950s, differences between sites were similar to those observed for the subsequent period, with the exception of the lower FOR subsection. At the oxbows, BD decreased slightly and whereas grain-size composition values were similar. For all the study sites, TOC and TN declined only minimally, whereas RIC average and variability increased at OL2, compared with the subsequent period. However, the deepest FOR sub-section, characterized by the absence of fines, exhibited a two-fold decrease in TOC and TN compared with the sediment higher in the profile. Despite this, OMI values did not differ significantly between both FOR subsections deposited in the early 1950s–1963 period, whereas C:N values slightly decreased.

Discussion

Spatio-temporal heterogeneity of C and N accumulation patterns

C and N accretion patterns within the study reach have been severely modified during the last century. Anthropogenic impacts have reduced the importance of allochthonous OM inputs. These changes have reduced the potential of the middle Ebro floodplains to act as C and N sinks because this floodplain function is maximized when fine sediment deposition (< 2 mm) is highest (Table 1). For the connected ecotope (BANK), the sedimentation rate was two-fold lower than that for FOR at the beginning of the period considered, when this habitat was thought to be at comparable successional stage (Fig. 1). Despite this, C and N accretion since 1963 was as high at BANK as in the connected oxbow lake. For the less connected ecotopes (OL2, OL3 and FOR), the most recently deposited sediment (h = 30 cm) did not seem to be affected by inputs of inorganic river seston (Fig. 4), whereas TOC and TN stocks were markedly higher after 1963 (Table 2). In addition, OMI increased markedly in the upper part of the profiles (Fig. 6), highlighting the inverse relationship between external inputs and organic matter incorporation in floodplain substrates (Rostan et al. 1987; Schwarz et al. 1996; Tockner and Schiemer 1997; Daniels 2003; Mitsch et al. 2005). From a quality standpoint, this shift in favour of OM inputs was not accompanied by any significant change in the size of the refractory OM pool. However, the greater degree of variability observed prior to 1963 reflected the dominance of allochthonous inputs (Table 2, Fig. 5). At more connected stages, the quality of particulate and sediment-associated OM was less uniform because of deposition over the surface during floods. Due to variability at the source, allochthonous OM inputs might have been more variable if compared with in situ-produced OM, which is thought to be more homogenous in terms of quality.

Spatial heterogeneity in accretion patterns could be accounted for by differences in successional pathways at the study sites that fostered distinct geomorphological traits during the sub-periods examined. Prior to the 1950s, dynamic geomorphological processes prevailed, as reflected by the migration of the main channel between 1927 and 1946, the extent of the area covered by gravels (Fig. 1) and the elevated rates of sediment accretion (Table 1). This situation promoted homogeneity of the C and N stocks (Table 2 and Fig. 4), due to the dominance of external inputs. Although oxbow lakes exhibited higher TOC and TN concentrations, similar C and N stocks were found, because of the higher bulk density of the FOR substrate. However, a fuller analysis of the entire OL2 and OL3 profiles down to the gravel bed (i.e., 1927) would be necessary to assess the effect of the different sedimentation environments (as interpreted from grain-size profiles and site geomorphology) on C and N accumulation rates. At FOR, there was an abrupt shift from a highly connected state, where conditions did not allow fines to settle, towards a lower-energy environment (Fig.4). Differences in clay, fines, and bulk density (Table 2 and Fig. 4) between FOR and the oxbow lake sites might reflect differences between the intermittently flooded FOR habitat, with its higher scouring potential and absence of water during low-discharge periods, and the greater water-residence time and inundation associated with ordinary floods, characteristic of the oxbow lakes.

Between the early 1950s and 1963, the dominance of external inputs and the homogeneity of C and N accretion patterns that prevailed within the study reach prior to the 1950s persisted at the oxbows, but at FOR, incorporation of in situ-produced OM seemed to be important for sediment deposited toward the end of this period (Fig. 6). Because of this, and because OM bulk density was higher, the OM stock was slightly higher at FOR. This could have been caused by a decrease in the frequency of inundation, which would have led to the establishment of mature riparian communities and a reduction in OM exports to the main channel (Mitsch and Gosselink 1993; Naiman and Decamps 1997; Robertson et al. 1999; Corenblit et al. 2007). This process could explain the coarser grain size compared with the previous period and might indicate that sedimentation occurred during low-frequency floods (Asselman and Middlekoop 1998). However, it could also be caused by incorporation of recently produced OM through leaching from surface layers or below-ground biomass. At OL2, the construction of a dyke at its upstream end between 1946 and 1957 (Ollero 1995) probably caused the hydroperiod of OL2 to be determined primarily by backflow inputs, leading to the dominance of fines (Table 2 and Fig. 6).

Since 1963, anthropogenic influences reduced the overall hydrological connectivity between the main channel and the floodplain. The coarse organic substrates in the upper part of the profiles associated with mature ecotopes might be a reflection of these changes (Table 2, Figs. 3 and 5). This substrate profile could also reflect the fact that only low-frequency floods with a higher proportion of sands affected these habitats, which would have become OM-enriched during the prolonged periods between floods. Also, redistribution from surrounding habitats subjected to higher sedimentation rates could have introduced coarser and OM-enriched sediment over the depression areas where cores were taken. At FOR, autogenic dominance occurred throughout the entire period, leading to maximum OM enrichment (Table 2). C and N stocks were two-fold higher compared with oxbow lakes, probably because rarely flooded habitats had more dense substrates. Slightly higher RIC values at FOR (Table 3) could point to differences in OM quality between terrestrial and aquatic communities, the latter being slightly more labile (Hein et al. 2003; Kosche 2006). At the oxbows, patterns shifted progressively after 1963, although an event at approximately the 30-cm depth appeared to have led to the current disconnected state of these habitats. TOC and TN concentrations significantly decreased during this event, which deposited fine inorganic river seston (Figs. 4 and 5). C and N budgets were higher at OL2 than at OL3, probably because of its higher connectivity with the main channel and alluvial aquifer, which are both nitrate enriched (Torrecilla et al. 2005). Higher nutrient availability has been previously observed to increase C and N accretion (Craft and Richardson 1998, Rybczyk et al. 2002; Wang et al. 2004), which would also account for the very high TOC, TON and OMI values found in the upper OL2 profile (Figs. 4 and 5). In terms of OM quality, RIC appeared to increase with OMI in the oxbow lakes (Fig. 3), which could have been caused by the deposition of fresh organic compounds over the surface (Tan et al. 2004). A more detailed analysis is required to determine whether differences in the structure of the bank communities at OL3 (Cabezas et al. in press) caused its lower RIC values (Table 2).

Potential to serve as C and N sinks

The middle Ebro floodplains have a high capacity to accumulate C and N, especially when hydrological connectivity with the main channel is sufficiently high to ensure the dominance of external inputs. After 1963, C and N accretion rates (Table 1) were in the highest range for those reported in freshwater marshes in the USA (see Wang et al. 2004), and clearly exceeded those during the previous decades when allochthonous inputs dominated. For a system whose function had been so heavily modified, C accretion (219.45 g C m^{-2} y^{-1}) was similar to the overall average rate (210 g C m^{-2} y^{-1}) calculated by Chmura et al. (2003) for mangroves and salt marshes (n = 26) worldwide. Under more pristine conditions, this C accretion capacity was almost two-fold higher in the middle Ebro floodplains (378.46 g C m^{-2} y^{-1}). Even at the lower current levels, the potential of floodplain substrates to act as C sinks is more than an order of magnitude greater than has been reported for peatlands (20–30 g C m^{-2} y^{-1}) in Canada (Roulet et al. 2000), which have received considerable attention in this context because of their vast expanse. Moreover, C accumulation in soil due to the conversion of agricultural fields into forest and grasslands is also ten-fold smaller (Post and Known 2000) than the averages reported in this study, coinciding with rates calculated by Schlesinger (1990) for 40- to 50-y-old soils.

Our results indicate that accumulation of recalcitrant OM, expressed as ROC in this paper, was more dependent on the total amount of carbon than its biochemical complexity, as has been reported elsewhere for temperate soils (Trumbore 1993; Paul et al. 2006). Similar trends were found for ROC and TOC accretion rates (Table 1) because average RIC values, which accounted for approximately half the total carbon stock, were similar for different sub-sections (Table 2). With regards to OM processing over time, this appeared to be reflected in our results, which suggested a modification of the original sediment characteristics since deposition. At OL2, OL3 and FOR, the proportion of fine sediment (< 63 μm) did not appear to affect TOC or TN concentrations for either period examined (Fig. 3), a conclusion that runs counter to the prevailing floodplain sedimentation paradigm (Asselman and Middlekoop 1995; Walling and He 1997; Steiger and Gurnell 2003, Noe and Hupp 2005). Our observation also explains the positive correlation between OMI and sand content for the upper part of the profiles (Fig. 6), because fines did not cause higher TOC or TN concentrations, as assumed in the OMI formula. OM depletion due to processing is thought to have caused this trend for sediment deposited before 1963. The low and relatively constant TOC and TN during this period support this idea (Table 2 and Fig. 4). For material deposited after 1963 at OL2, OL3 and FOR, the lack of correlation between in situ-produced OM and grain size could explain our results. Although the TOC concentration in the last FOR sub-section, which was lacking in fines, appears to be inconsistent with our assumptions, this discrepancy could be due to the preferential groundwater flow made possible by the coarser grain-size at this site, which would promote greater dissolved-OM exports to the main channel. At BANK, where the sediment has been subjected to processing for a shorter period of time, TOC and TN concentrations appeared to increase with the proportion of fines (Fig. 3) and OMI was seemingly unaffected by grain size (Fig. 6).

Changes that occurred with processing are not reflected in the OM quality variables included in this study. The C:N ratio is a useful indicator with predictive value in litter decomposition studies (Nicholardot et al. 2001), but it has been found to vary differently between different SOM fractions (Kaye et al. 2003; Tan et al. 2004), which were not determined in this study. With respect to RIC, Rovira and Vallejo (2002) found using H_2SO_4 hydrolysis that RIC decreased in plant tissues as decomposition progressed, although they did not observe any relationship using HCl hydrolysis. Others have found that RIC decreased with depth where SOC was older and, therefore, more processed (Tan et al. 2004; Rovira and Vallejo 2007). Indeed, new approaches have questioned the paradigm that labile SOM decreases with decomposition (Rovira et al. 2008), arguing that future studies should employ more modern methods. In addition, the role of physical protection (Mikkuta et al. 2006) in modifying the potential of floodplain substrates to act as OM sinks should be considered.

Implications for restoration and management

Our results suggest that it would be appropriate to consider the potential role of floodplains as C and N sinks within the context of future Ebro basin management strategies. As indicated by Owens et al. (2005), the management of fine-grained sediment dynamics by ensuring interconnectivity between the different basin compartments is an urgent priority. At the reach scale, the first important task is the conversion of human-occupied areas, which currently cover most of the floodplain (Fig.1), into natural ecotopes. The harvesting of agricultural fields and poplar groves results in lower incorporation of autochthonous OM into the superficial substrate than occurs in mature habitats (unpublished data). Allochthonous OM inputs are also lower at these sites than at connected ecotopes, because they remain isolated during floods. Secondly, the geomorphological dynamic should be restored to the extent possible, so as to enlarge the proportion of early stage ecotopes. Currently, the area covered by natural ecotopes is dominated by mature stages (Fig. 1), resulting in a severe restriction in the area dominated by allochthonous inputs, where C and N accumulation is higher. This target would be synergistic with the related restoration targets linked to water quality enhancement and maintenance of a shifting-habitat mosaic. Moreover, the hydroperiod for mature ecotopes should be managed and maintained so as to avoid terrestrialization. In situ-produced organic matter at these ecotopes might be greater than at adjacent upland patches because of the semi-arid climate of the region, where annual average precipitation is less than 300 mm.

To successfully implement those goals, the natural flow regime should be restored. As suggested by Comin (1999), alternative strategies would need to make more integrated use of natural resources, primarily soil and water. At the reach scale, re-establishing sediment exchange and natural hydroperiods would require periodic economic investment to restructure dikes or lower floodplain heights, as suggested for other European rivers (Schiemer et al. 1999; Hughes et al. 2001; Buijse et al. 2002). However, benefits may extend to the global scale, in as far as ecosystem restoration will contribute to the accumulation of carbon and nitrogen in floodplains, as has been shown in this paper.

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REFERENCES

Amoros C, Bornette G 2002. Connectivity and biocomplexity in waterbodies of riverine floodplains. Freshwater Biol 47: 761-776.

Amoros C, Wade PM 1996 Ecological succesions. In: Fluvial Hydrosystems, Petts GE, Amoros C (eds) Chapman & Hall, London. pp. 211-241

Asselman NEM, Middelkoop H 1995. Floodplain Sedimentation - Quantities, Patterns and Processes. Earth Surf Processes 20: 481-499.

Asselman NEM, Middelkoop H 1998. Temporal variability of contemporary floodplain sedimentation in the Rhine-Meuse Delta, the Netherlands. Earth Surf Processes 23: 595-609.

Baptist MJ, Penning WE, Duel H, Smits AJM, Geerling GW, van der Lee GEM, Van Alphen JSL 2004. Assessment of the effects of cyclic floodplain rejuvenation on flood levels and biodiversity along the Rhine river. River Res Appl 20: 285-297.

Batalla RJ, Gomez CM, Kondolf GM 2004. Reservoir-induced hydrological changes in the Ebro River basin (NE Spain). J Hydrol 290: 117-136.

Begueria S, Lopez-Moreno JI, Gomez-Villar A, Rubio V, Lana-Renault N, Garcia-Ruiz JM 2006. Fluvial adjustments to soil erosion and plant cover changes in the Central Spanish Pyrenees. Geogr Ann Ser a-Phys Geogr 88A: 177-186.

Begueria S, Lopez-Moreno JI, Lorente A, Seeger M, Garcia-Ruiz JM. 2003. Assessing the effect of climate oscillations and land-use changes on streamflow in the Central Spanish Pyrenees Ambio 32: 283-286.

Buijse AD, Coops H, Staras M, Jans LH, Van Geest GJ, Grift RE, Ibelings BW, Oosterberg W, Roozen FCJM 2002. Restoration strategies for river floodplains along large lowland rivers in Europe. Freshwater Biol 47: 889-907.

Cabezas A, Gonzalez E, Gallardo B, García M, Gonzalez M, Comín FA in press. Effects of hydrological connectivity on sediment and understory structure in the middle Ebro river (NE Spain): Implications for restoration and management. Aquat Sci.

Campbell BL, Loughran TJ, Elliot GL 1988 A method for determining sediment budgets using caesium 137. In In Sediment Budgets, Proceedings of the Porto Alegre Symposium, December 1988, IAHS Publication. 171-179. (IAHS press, Institute of Hydrology: Wallingford)

Chmura GL, Anisfeld SC, Cahoon DR, Lynch JC 2003. Global carbon sequestration in tidal, saline wetland soils. Glob Biogeochem Cycles 17.

Comin FA 1999. Management of the Ebro River Basin: Past, present and future. Water Sci Technol 40: 161-168.

Corenblit D, Tabacchi E, Steiger J, Gurnell AM 2007. Reciprocal interactions and adjustments between fluvial landforms and vegetation dynamics in river corridors: A review of complementary approaches. Earth-Sci Rev 84: 56-86.

Craft CB, Richardson CJ 1998. Recent and long-term organic soil accretion and nutrient accumulation in the everglades. Soil Sci Soil Am J 62: 834-843.

Daniels JM 2003. Floodplain aggradation and pedogenesis in a semiarid environment. Geomorphol 56: 225-242.

Day JW, Ko JY, Rybczyk J, Sabins D, Bean R, Berthelot G, Brantley C, Cardoch L, Conner W, Day JN, Englande AJ, Feagley S, Hyfield E, Lane R, Lindsey J, Mistich J, Reyes E, Twilley R 2004. The use of wetlands in the Mississippi Delta for wastewater assimilation: a review. Ocean Coast Manag 47: 671-691.

Garcia-ruiz JM, Lasanta T, Ortigosa L, Ruizflano P, Marti C, Gonzalez C 1995. Sediment yield under different land uses in the spanish pyrenees. Mt Res Dev 15: 229-240.

Hein T, Baranyi C, Herndl GJ, Wanek W, Schiemer F 2003. Allochthonous and autochthonous particulate organic matter in floodplains of the River Danube: the importance of hydrological connectivity. Freshwater Biol 48: 220-232.

Hughes FMR, Adams WM, Muller E, Nilsson C, Richards KS, Barsoum N, Decamps H, Foussadier R, Girel J, Guilloy H, Hayes A, Johansson M, Lambs L, Pautou G, Peiry JL, Perrow M, Vautier F, Winfield M 2001. The importance of different scale processes for the restoration of floodplain woodlands. Regul Rivers-Res Manag 17: 325-345.

Ibanez C, Prat N, Canicio A 1996. Changes in the hydrology and sediment transport produced by large dams on the lower Ebro river and its estuary. Regul Rivers-Res Manag 12: 51-62.
IPCC. 2000 Land Use, Land Use Change, and Forestry. Cambridge University Press, Cambridge. United Kingdom.
Johnston CA 1991. Sediment and nutrient retention by freshwater wetlands- Effects on surface-water quality. Crit Rev Environ Control 21: 491-565.
Junk WJ, Bayley B, Sparks RE 1989. The flood pulse concept in river-floodplain systems. In: D.P.Dodge (ed) Proc. Intern. Large River Symposium. Special Issue of J.Canad. Fish and Aquat. Sci. 106: 11-27.
Kaye JP, Binkley D, Rhoades C 2003. Stable soil nitrogen accumulation and flexible organic matter stoichiometry during primary floodplain succession. Biogeochem 63: 1-22.
Knosche R 2006. Organic sediment nutrient concentrations and their relationship with the hydrological connectivity of floodplain waters (River Havel, NE Germany). Hydrobiologia 560: 63-76.
Lopez-Moreno JI, Begueria S, Garcia-Ruiz JM 2006. Trends in high flows in the central Spanish Pyrenees: response to climatic factors or to land-use change? Hydrol Sci J 51: 1039-1050.
McCarty GW, Ritchie JC 2002. Impact of soil movement on carbon sequestration in agricultural ecosystems. Env Pollut 116: 423-430.
Mikutta R, Kleber M, Torn MS, Jahn R 2006. Stabilization of soil organic matter: Association with minerals or chemical recalcitrance? Biogeochem 77: 25-56.
Mitsch WJ, Gosselink JG 1993 Wetlands. Van Nostrand Reinhold, New York.
Mitsch WJ, Zhang L, Anderson CJ, Altor AE, Hernandez ME 2005. Creating riverine wetlands: Ecological succession, nutrient retention, and pulsing effects. Ecol Eng 25: 510-527.
Naiman RJ, Decamps H 1997. The ecology of interfaces: Riparian zones. Annual Review of Ecology and Systematics 28: 621-658.
Nicolardot B, Recous S, Mary B 2001. Simulation of C and N mineralisation during crop residue decomposition: A simple dynamic model based on the C : N ratio of the residues. Plant and Soil 228: 83-103.
Noe GB, Hupp CR 2005. Carbon, nitrogen, and phosphorus accumulation in floodplains of Atlantic Coastal Plain rivers, USA. Ecol Appl 15: 1178-1190.
Ollero A 1992. Los meandros libres de el Ebro (Logroño-La Zaida). Geomorfología fluvial, ecogeografía y Riesgos. Tesis Doctoral, Zaragoza, Zaragoza.
Ollero A 1995. Dinámica reciente del cauce de el Ebro en la Reserva Natural de los Galachos. Cuater Geomorphol 9: 85-93.
Owens PN, Batalla RJ, Collins AJ, Gomez B, Hicks DM, Horowitz AJ, Kondolf GM, Marden M, Page MJ, Peacock DH, Petticrew EL, Salomons W, Trustrum NA 2005. Fine-grained sediment in river systems: Environmental significance and management issues. River Res Appl 21: 693-717.
Paul EA, Morris SJ, Conant RT, Plante AF 2006. Does the acid hydrolysis-incubation method measure meaningful soil organic carbon pools? Soil Sci Soil Am J 70: 1023-1035.
Poff NL, Allan JD, Bain MB, Karr JR, Prestegaard KL, Richter BD, Sparks RE, Stromberg JC 1997. The natural flow regime. Biosci 47: 769-784.
Post WM, Kwon KC 2000. Soil carbon sequestration and land-use change: processes and potential. Glob Chan Biol 6: 317-327.
Ritchie JC, McHenry JR 1990. Application of radioactive fallout cesium-137 for measuring soil-erosion and sediment accumulation rates and patterns - a review. J Environ Qual 19: 215-233.
Robertson AI, Bunn SE, Boon PI, Walker KF 1999. Sources, sinks and transformations of organic carbon in Australian floodplain rivers. Marine and Freshwater Res 50: 813-829.
Rostan JC, Amoros C, Juget J 1987. The Organic Content of the Surficial Sediment - a Method for the Study of Ecosystems Development in Abandoned River Channels. Hydrobiologia 148: 45-62.
Roulet NT 2000. Peatlands, carbon storage, greenhouse gases, and the Kyoto Protocol: Prospects and significance for Canada. Wetlands 20: 605-615.
Rovira P, Kurz-Besson C, Couteaux MM, Vallejo VR 2008. Changes in litter properties during decomposition: A study by differential thermogravimetry and scanning calorimetry. Soil Biol Biochem 40: 172-185.
Rovira P, Vallejo VR 2000. Examination of thermal and acid hydrolysis procedures in characterization of soil organic matter. Commun Soi Sci Plant Anal 31: 81-100.
Rovira P, Vallejo VR 2002. Labile and recalcitrant pools of carbon and nitrogen in organic matter decomposing at different depths in soil: an acid hydrolysis approach. Geoderma 107: 109-141.
Rovira P, Vallejo VR 2007. Labile, recalcitrant, and inert organic matter in Mediterranean forest soils. Soil Biol Biochem 39: 202-215.
Rybczyk JM, Day JW, Conner WH 2002. The impact of wastewater effluent on accretion and

decomposition in a subsiding forested wetland. Wetlands 22: 18-32.
Schiemer F, Baumgartner C, Tockner K 1999. Restoration of floodplain rivers: The 'Danube restoration project'. Regul Rivers-Res Manag 15: 231-244.
Schlesinger WH 1990. Evidence from chronosequence studies for a low carbon-storage potential of soils. Nature 348: 232-234.
Schwarz WL, Malanson GP, Weirich FH 1996. Effect of landscape position on the sediment chemistry of abandoned-channel wetlands. Landscape Ecol 11: 27-38.
Steiger J, Gurnell AM 2003. Spatial hydrogeomorphological influences on sediment and nutrient deposition in riparian zones: observations from the Garonne River, France. Geomorphol 49: 1-23.
Steiger J, Tabacchi E, Dufour S, Corenblit D, Peiry JL 2005. Hydrogeomorphic processes affecting riparian habitat within alluvial channel-floodplain river systems: A review for the temperate zone. River Res Appl 21: 719-737.
Tan ZX, Lal R, Izaurralde RC, Post WM 2004. Biochemically protected soil organic carbon at the north appalachian experimental watershed. Soil Sci 169: 423-433.
Thoms MC 2003. Floodplain-river ecosystems: lateral connections and the implications of human interference. Geomorphol 56: 335-349.
Tockner K, Pennetzdorfer D, Reiner N, Schiemer F, Ward JV 1999. Hydrological connectivity, and the exchange of organic matter and nutrients in a dynamic river-floodplain system (Danube, Austria). Freshwater Biol 41: 521-535.
Tockner K, Schiemer F 1997. Ecological aspects of the restoration strategy for a river-floodplain system on the Danube River in Austria. Glob Ecol Biogeograph Lett 6: 321-329.
Torrecilla NJ, Galve JP, Zaera LG, Retarnar JF, Alvarez A 2005. Nutrient sources and dynamics in a mediterranean fluvial regime (Ebro river, NE Spain) and their implications for water management. J Hydrol 304: 166-182.
Trumbore SE 1993. Comparison of Carbon Dynamics in Tropical and Temperate Soils Using Radiocarbon Measurements. Glob Biogeochem Cycles 7: 275-290.
Verhoeven JTA, Arheimer B, Yin CQ, Hefting MM 2006. Regional and global concerns over wetlands and water quality. Trends Ecol Evol 21: 96-103.
Vericat D, Batalla RJ 2006. Sediment transport in a large impounded river: The lower Ebro, NE Iberian Peninsula. Geomorphol 79: 72-92.
Walling DE, Bradley SB 1990. Some applications of caesium-137 measurements in the study of erosion, transport and deposition. . In: Erosion, Transport and depostion procceses, Proceedeings of the Jerusalem workshop (March-April 1987) IAHS publication. IAHS Press, Institute of Hydrology: Wallingford pp. 179-203.
Walling DE, He Q 1997. Investigating spatial patterns of overbank sedimentation on river floodplains. Water Air Soil Poll 99: 9-20.
Walling DE, He Q, Nicholas AP 1996 Floodplains as Suspended Sediment Sinks. In: Floodplain Processes, Anderson GM, Walling DE, Bates PD (eds). Wiley, Chichester. pp 399-440.
Walling DE, Owens PN, Carter J, Leeks GJL, Lewis S, Meharg AA, Wright J. 2003. Storage of sediment-associated nutrients and contaminants in river channel and floodplain systems. Appl Geochem 18: 195-220.
Wang GP, Liu JS, Tang H. 2004. The long-term nutrient accumulation with respect to anthropogenic impacts in the sediments from two freshwater marshes (Xianghai Wetlands, Northeast China). Water Res 38: 4462-4474.
Ward JV, Stanford JA. 1995. Ecological Connectivity in Alluvial River Ecosystems and Its Disruption by Flow Regulation. Regul Rivers-Res Manag 11: 105-119.
Ward JV, Tockner K, Uehlinger U, Malard F. 2001. Understanding natural patterns and processes in river corridors as the basis for effective river restoration. Regul Rivers-Res Manag 17: 311-324.

Conclusions and further research

1. The river disturbance regime is the primary driver of floodplain structure and organic matter accretion patterns in the Middle Ebro River.

2. Direct (modification of the natural flow regime) and indirect (flood protection structures and occupation of the river space) human disruptions have greatly modified the river disturbance regime, and so river-floodplain interactions responsible of maintaining the ecological integrity of the Middle Ebro floodplains.

3. As a result, both the structure of the different components examined in this thesis (landscape structure, hydrogeochemistry, sediment structure, riparian understory) and the organic matter accretion have been deviated from a more natural patterns.

4. Such alterations exerted a direct effect over the riverine landscape, promoting the dominance of mature stages of landscape ecotopes and riparian wetlands. Meanwhile, the exchange of mass, energy and organism (hydrological connectivity) between the different compartments which composed the river-floodplain system has been restricted to a lesser extent during shorter time-spans.

5. Landscape diversity has drastically decreased since the 60´s. Mature ecotopes have became dominant since that, whereas ecotope rejuvenation only takes place within the main channel. To increase the landscape heterogeneity, both hydrological and landscape factors must be considered.

6. Analogously, the riverscape at the study reach (array of riparian wetlands) is homogeneous, in terms of riparian succession, due to the predominance of mature successional stages.

7. The hydrogeochemical characteristics of surface water bodies at the study reach are explained by the inputs from different compartments (main channel, hyporheic, alluvial aquifer). It promotes a gradient of mineralization and fertilization in the surface waters. In future studies, it is required to quantify the discharge from the different compartments and the rate of different biogeochemical processes.

8. Hydroperiod and flooded sediment structure of riparian wetlands were closely related at the study reach. At superficially connected wetlands, inorganic sediment inputs characterize the sediment composition. In turn, at superficially disconnected wetlands, in situ produced organic matter is incorporated to the substrate increasing the organic matter budgets.

9. A relatively high diversity (in terms of hydrogeochemistry, wetlands hydroperiod and flooded sediment structure) was found at the study reach. Given the current geomorphological conditions, we hypothesized that this heterogeneity will not remain at larger spatio-temporal scales.

10. The evolution of riparian landforms was valid to explain organic matter accretion patterns at the top-soil (0-10 cm) of riparian habitats. It could be used for a rapid assessment of the organic matter budgets at the top-soil of different river stretches.

11. At mature natural landforms (>60 y), autochtonous inputs dominate and carbon and nitrogen budgets at the top-soil (0-10 cm) are the highest. At young natural landforms (< 60 y), deposition or inorganic sediments during floods dominate and carbon and nitrogen budgets are lower, although similar to those found in

agricultural fields and poplar groves, which remain isolated during floods.

12. During the last century, C and N accretion patterns at the study reach have been modified. Anthropic influences have lowered the importance of alloctonous OM inputs. It reduced the potential of the Middle Ebro floodplains as C and N sinks because this function maximizes when fine sediment deposition (< 2 mm) dominate over endogenous inputs.

13. Even so, the middle Ebro floodplains have a high C and N accumulation capacity. Rates were in the highest range for other wetlands worldwide, exceeding those quantities in during the first decades of the 20^{th} century, when alloctonous inputs dominated. Anyway, the potential of floodplain substrates as C sinks is over one order of magnitude higher than this reported for peatlands or for conversion of agricultural fields to natural habitats.

14. According to our results, the refractory soil organic matter pool relied on the quantity the bulk pool rather than on its biochemical complexity, i.e quality. Further studies performing more detailed analyses should be implemented for a correct characterization of the protected soil organic matter pool. Also the physical protection exerted by the substrate should be analyzed.

15. To improve the ecological status of the study reach, ecological restoration must be implemented at basin, reach and site scale. Since self-sustained restoration is neither possible nor realistic under the current basin management, so periodic economic investments will be required.

16. We hypothesized that ecotope and wetland heterogeneity will increase after restoration. Also hydroperiod, sediment and hydrogeochemical diversity will expand to larger areas during low-frequency floods if rehabilitation of river-floodplain interactions is performed.

17. If any restoration project is performed, monitoring the effect over floodplain components examined in this thesis is highly desirable. This is the only way to asses if restoration guidelines proposed along this thesis are valid.

18. Also the evaluation of carbon and nitrogen accretion rates after restoration should be afforded in future studies. By quantifying those budgets, we should be able to valuate one of the benefits of ecological restoration.

19. Carbon and Nitrogen biogeochemical cycles should be analyzed and modelled in future studies. The role of riverine floodplains as buffer strips for agricultural runoff could therefore be assessed, as well as the balance of greenhouse gasses (nitrous oxide, methane and carbon dioxide), which belongs from the carbon and nitrogen processing.

Conclusiones y futuros objetivos de investigación

1. El régimen de alteración fluvial es el principal factor a la hora de explicar la estructura y la acumulación de materia orgánica en las llanuras de inundación del Ebro Medio.

2. Las alteraciones directas (modificación del régimen de caudales natural) e indirectas (estructuras de protección contra las avenidas y ocupación del estado fluvial) han modificado de manera muy notable el régimen de alteración fluvial, y por lo tanto las interacciones entre el río y las llanuras de inundación, que mantienen la integridad ecológica en las llanuras de información del Ebro Medio.

3. Como resultado, la estructura de las diferentes componentes que se han analizado en esta tesis (paisaje, hidrogeoquímica, estructura del sedimento, sotobosque ripario) difiere de unos patrones más naturales, así como la acumulación de materia orgánica en el sustrato.

4. Estas alteraciones han afectado directamente al paisaje fluvial, promoviendo el dominio de estadíos maduros, tanto de ecotopos como de humedales riparios. Al mismo tiempo, el intercambio de materia, energía y organismos (conectividad hidrológica) entre los diferentes compartimentos que componen el sistema río-llanura de inundación se ha restringido a áreas más pequeñas durante intervalos de tiempo más cortos.

5. La diversidad de paisaje ha descendido drásticamente desde los años 60. La proporción de la llanura de inundación dominada por los ecotopos más maduros ha aumentado desde entonces, mientras que la renovación solo tiene lugar dentro del cauce del río. Para aumentar la heterogeneidad del paisaje, los patrones hidrológicos y de paisaje actuales deben de ser considerados.

6. Asimismo, el conjunto de humedales riparios de nuestro tramo de estudio es homogéneo, en términos sucesión riparia, y se encuentra dominado por estadíos maduros.

7. La características hidrogeoquímicas de los cuerpos de agua superficiales en el tramo de estudio es el resultado de las aportaciones de los diferentes compartimentos (aportes superficiales del río Ebro, hiporréico, acuífero aluvial). Esto ocasiona un gradiente de mineralización y fertilización en el conjunto de las aguas superficiales. En estudios futuros, será necesario cuantificar las aportaciones de cada uno de los compartimentos y las tasas de diferentes procesos biogeoquímicos.

8. El hidroperiodo y la estructura del sedimento están íntimamente relacionados en los humedales riparios del área de estudio. En humedales conectados con el río Ebro de manera superficial, los aportes de sedimentos inorgánicos caracterizan la composición del sedimento. Por el contrario, en humedales desconectados superficialmente, la material orgánica producida in-situ se incorpora al sustrato incrementando su contenido en materia orgánica.

9. En el tramo de estudio, se encontró una diversidad relativamente alta cuando la hidrogeoquímica, el hidroperiodo y la estructura de los sedimentos inundados fueron analizados. Sin embargo, esta heterogeneidad desaparece cuando se tienen en cuenta escalas espacio-temporales mas amplias.

10. La evolución geomorfológica de los hábitats riparios fue útil para explicar los

patrones de acumulación de materia orgánica en el sustrato superficial (0-10 cm.) de susodichos hábitats. Esto puede ser empleado para una evaluación rápida de las cantidades de materia orgánica acumuladas en los sustratos superficiales de otros tramos de río.

11. En los hábitats maduros, (>60 años), los aportes endógenos predominan, y las cantidades de carbono y nitrógeno en el sustrato superficial (0-10 cm) son las más grandes. En hábitats más jóvenes (< 60 años), el aporte de sedimentos inorgánicos durante las riadas predomina, y las cantidades de carbono y nitrógeno son las mas bajas, aunque similares a las que se han estimado para los sustratos superficiales de campos de cultivo y choperas, que permanecen aislados durante las riadas.

12. Durante el último siglo, los patrones de acumulación de carbono y nitrógeno en el área de estudio han sido modificados. Las alteraciones antrópicas han reducido la importancia de los aportes exógenos. Por este motivo, se ha reducido el potencial de las llanuras de inundación del Ebro Medio como sumideros de carbono y nitrógeno ya que esta función se maximiza cuando la sedimentación de sedimentos finos (< 2 mm) domina sobre los aportes endógenos.

13. A pesar de las alteraciones antrópicas, las llanuras de inundación del Ebro Medio tienen una gran capacidad de acumulación de carbono y nitrógeno. Las tasas calculadas son similares a las máximas estimadas para distintos humedales en otras partes del mundo, superándolas en la primera parte del siglo XX, cuando los aportes exógenos dominaban. Además, el potencial supera en un orden de magnitud a las tasas descritas para turberas o por la re-naturalización de campos de cultivo.

14. Según nuestros resultados, la cantidad de material orgánica considerada como refractaria depende más de la cantidad total de materia orgánica del sustrato que de la protección bioquímica, i.e. calidad. En futuras investigaciones, se debería de realizar estudios más detallados para caracterizar de una manera mas completa el "pool" de materia orgánica protegida. El papel de la protección física que el sustrato ejerce sobre la materia orgánica también debería de ser analizado.

15. Para mejorar el estado ecológico del tramo de estudio, la restauración ecológica debería de ser implementada a escala de cuenca, tramo y hábitat. Con la gestión actual de la cuenca, una restauración auto-mantenida no es posible ni realista, por lo tanto, se requerirá de una inversión económica periódica.

16. Nuestra hipótesis es que la heterogeneidad de ecotopos y humedales riparios incrementará después de la restauración. De la misma manera, la diversidad de hidroperiodos, hidrogeoquímica y de sedimentos inundados se extenderá a áreas mas grandes y durante riadas de mayor magnitud, siempre que se rehabiliten las interacciones entre el río y las llanuras de inundación.

17. Si algún proyecto de restauración es ejecutado, el monitoreo del efecto sobre los componentes del sistema incluidos en este estudio es altamente recomendable. Solo así podremos saber si las líneas de actuación que se sugieren en esta tesis tienen validez.

18. Asimismo, se debería de evaluar en futuros estudios las tasas de acumulación de carbono y nitrógeno después de la restauración. A través de la cuantificación de estas tasas, se podría valorar económicamente uno de los beneficios de la restauración ecológica.

19. El ciclo biogeoquímico del carbono y el nitrógeno debería también ser objeto de estudio, así como el modelado de diferentes procesos incluidos en estos ciclos. El papel de las llanuras de inundación como zonas de atenuación de la contaminación agrícola difusa debe de ser evaluado, así como el balance de gases con efecto invernadero (oxido nitroso, metano o dióxido de carbono), los cuales derivan del procesado del carbono y del nitrógeno.