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Recent Changes in the Riparian Forest of a Large Regulated Mediterranean River: Implications for Management

Eduardo González · María González-Sanchis ·
Álvaro Cabezas · Francisco A. Comín ·
Etienne Muller

Abstract The structure of the floodplain forests of the Middle Ebro River (NE Spain) was examined at patch and landscape scales along a three-step chronosequence defined according to the extent of flow regulation-induced hydrogeomorphic changes, with the ultimate purpose of producing baseline information to guide through management and restoration plans. At patch scale, a total of 6,891 stems within 39 plots were registered for species, diameter and health status. The stem density, size class distribution, canopy dieback and mortality were further compared by means of non-parametric tests. At landscape scale, the temporal evolution of the area occupied by forest stands of different ages in the floodplain along the chronosequence was evaluated using four sets of aerial photographs dated in 1927, 1957, 1981 and 2003. The within-patch structure of *pioneer* forests (<25–30 years old) was characterized by dense and healthy populations of pioneer species (*Populus nigra*, *Salix alba* and *Tamarix* spp.), but the area occupied by these forest types has progressively decreased (up to 37%) since the intensification of river regulation (ca. 1957). In contrast, *non-pioneer* forests (>25–30 years old) were

characterized by declining and sparse *P. nigra*–*S. alba*–*Tamarix* spp. stands, where late-seral species such as *Ulmus minor* and *Fraxinus angustifolia* were frequent, but only as small-size stems. At landscape scale, these type of senescent forests have doubled their surface after river regulation was intensified. *Populus alba* only appeared in the oldest plots recorded (colonized before 1957), suggesting sexual regeneration failure during the last five decades, but usually as healthy and dense stands. Based on these findings, measures principally aimed at recovering some hydrogeomorphic dynamism are recommended to guarantee the self-sustainability of the floodplain forest ecosystem.

Keywords Floodplain forests · River regulation · Middle Ebro River · *Salicaceae* · *Tamarix* · Canopy dieback · Stand structure · Hydrogeomorphic · Mediterranean river

Introduction

The variety of riparian plant communities found in natural floodplains is mainly controlled by the flow regime (Poff and others 1997), which generates physical disturbance and environmental stress on riparian vegetation, ultimately affecting its temporal and spatial dynamics (Shafroth and others 2002). The effect of flow regime on riparian vegetation is hydrologic and geomorphologic (Bendix and Hupp 2000). On the one hand, hydrology influences the regeneration and maintenance of floodplain vegetation by transporting propagules, wetting floodplain soils, recharging ground-water and providing nutrients to the existing vegetation. On the other hand, fluvio-geomorphologic events create new bare substrates necessary for the establishment of pioneer plants (Mahoney and Rood 1998; Stromberg

E. González (✉) · M. González-Sanchis · Á. Cabezas ·
F. A. Comín
Pyrenean Institute of Ecology (IPE), Spanish National Research
Council (CSIC), Avda. Montañana 1005, 50192 Zaragoza, Spain
e-mail: edusargas@ipe.csic.es

E. Muller
Université de Toulouse, UPS, INP, EcoLab (Laboratoire
d'écologie fonctionnelle), 29 rue Jeanne Marvig,
31055 Toulouse, France

E. Muller
French National Center for Scientific Research (CNRS), EcoLab
(Laboratoire d'écologie fonctionnelle), 29 rue Jeanne Marvig,
31055 Toulouse, France

1998; Cooper and others 1999), which are progressively and sequentially succeeded by late-seral stages until eventually a severe flood brings the system back to a starting point (Johnson and others 1976; Schnitzler 1995; Ward and Stanford 1995; Cordes and others 1997; Stromberg and others 1997; Stromberg 1998; van Pelt and others 2006).

Flow regulation and flood protection infrastructures strongly modify the flow regime of rivers and, thereby, disrupt the hydrogeomorphic-dependent structure of plant communities (Busch and Smith 1995; Ward and Stanford 1995; Friedman and others 1996; Stromberg and others 1996; Poff and others 1997; Rood and others 1999; Gergel and others 2002; Shafroth and others 2002; Dufour and others 2007). Accordingly, it might be expected that the effects of regulation on the structure of the riparian tree community (i.e., the pattern of tree species composition, abundance and size class distribution) are reflected on within-patch forest metrics, such as stem density (Shafroth and others 2002). Other measurements, such as canopy dieback or stem mortality, might also give valuable information about the species fitness to the changing hydrologic conditions. Indeed, it was proposed by Rood and others (2000) that branch dieback in some species of *Populus* is auto-induced in drought stress scenarios, so that trees would reduce their water use. Horton and others (2001) showed that other riparian tree species besides *Populus*, namely *Salix gooddingii* and *Tamarix chinensis*, also sacrifice some branches in order to improve water status in the surviving shoot. Sometimes branch sacrifice is not enough to cope with dry conditions and water table depletions, and mortality can not be prevented. In fact, significant proportions of dead stems in stands can be interpreted as an evidence of severe hydrologic alteration (Megonigal and others 1997; Scott and others 2000).

Shifts in the structure of the riparian tree community after human-induced hydrogeomorphic changes have been largely studied in North America (Johnson and others 1976; Johnson 1992; 1998; Rood and others 1999; Bendix and Hupp 2000; Merritt and Cooper 2000; Gergel and others 2002; Shafroth and others 2002; Katz and others 2005) and Northwestern Europe (Marston and others 1995; Trémolières and others 1998; Dufour and others 2007). However, few such studies have been developed in the Mediterranean region (Crivelli and others 1995; Aguiar and others 2001; Salinas and Casas 2007). In Spain, although the hydrogeomorphic patterns of one of the largest rivers, namely the Ebro River (NE Iberian Peninsula), have dramatically changed during the last decades as a consequence of flow regulation and flood protection (Batalla and others 2004; Ollero 2007; Cabezas and others 2009), there has been no attempt to date to quantify how the structure of the riparian tree community has been affected by the progressive disruption of river-floodplain

interactions. To guide management plans, a better understanding of the relationships between the structure of riparian woodlands and their hydrogeomorphic drivers is required.

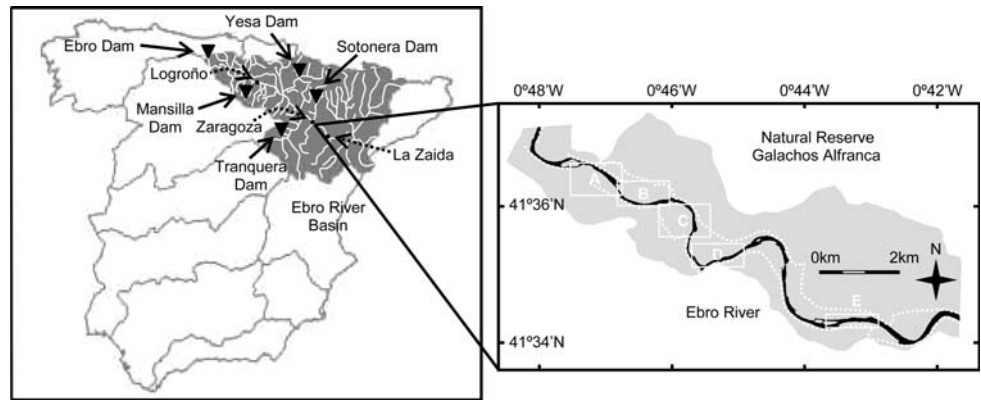
In this context, the main objective of this study is to describe the structure and health of the riparian tree community in the Middle Ebro River, within a framework of historic and recent hydrogeomorphic change. In particular, the composition, stem density, size class distribution, canopy dieback and mortality of the dominant tree species were expected to vary through an ~80-year chronosequence characterised by changing fluvio-geomorphic patterns that have resulted from river regulation. It was also hypothesized that the change of hydrogeomorphic dynamics would have influenced the relative proportions of forest stands of different ages along the chronosequence. The ultimate objective of this work is to provide background data useful for management and restoration decisions.

Methods

Study Area

The studied area is the Natural Reserve of the Alfranca Galachos (41°36'N, 0°46'W), an 8-km river segment located 12 km downstream of the city of Zaragoza in the Middle Ebro River, which is a 346-km reach between Logroño and La Zaida (NE Spain) (Fig. 1). The Ebro River is the second largest river in the Iberian Peninsula in terms of length (930 km), annual averaged discharge (~12,000 hm³ year⁻¹) and drainage area (85,534 km²). The average monthly discharge registered at the gauging station of Zaragoza is 230 m³ s⁻¹, and the elevation in the study reach ranges between 175 and 185 m above sea level (a.s.l.). The precipitation regime of the semiarid Ebro depression is characterized by rainfall scarcity (annual average ~400 mm), high inter- and intra-annual variability, with maximum levels in spring and autumn and minimum levels in summer (Pinilla 2006). The overstory of the natural floodplain forest is composed of three softwood *Salicaceae* species: white poplar (*Populus alba*), European black poplar (*P. nigra*) and white willow (*Salix alba*), and five hardwood species: saltcedar (*Tamarix gallica*, *T. africana* and *T. canariensis*), narrow leaf ash (*Fraxinus angustifolia*) and field elm (*Ulmus minor*). Other species, such as the shrubs common hawthorn (*Crataegus monogyna*) and bay laurel (*Laurus nobilis*) rarely attain tree size on the floodplain. The exotics box elder (*Acer negundo*), black locust (*Robinia pseudoacacia*), babylon willow (*Salix babylonica*) and some fruit trees have been occasionally detected as tree-size stems. Steppic shrubby vegetation dominates the uplands adjacent to the floodplain.

Fig. 1 Localization of the study area. The *grey zone* represents the area flooded during the 10-year flood of February 2003. *Dotted white lines* are flood defence structures



Hydrogeomorphic Background

The hydrogeomorphology of the Middle Ebro River during the twentieth century has evolved during three major periods following human interventions (Ollero 2007; Cabezas and others 2009):

(1) In a *pre-regulation* period (before the 1950s), the natural pluvionival hydrological regime of the Ebro River remained practically unaltered. In the absence of regulation, floods usually occurred in winter and spring, when precipitation was more frequent, and after the snowmelt. Low water levels normally lasted from July to October (Fig. 2a). The inter-annual variability of the flow regime and flood pulses was high (Fig. 2b). With an almost unmodified flow regime and absence of significant flood protection, the middle stretch of the river had a braided-meandering channel with unstable riverbanks. In this period, changes in the channel morphology (i.e., meander cut-offs produced during flood episodes and channel migration caused by progressive bank erosion) were continuous and, consequently, natural forests concentrated in the outer floodplain. However, agricultural practices started to convert extensive areas of the floodplain to anthropic landcover and to narrow the natural riparian corridor.

(2) *Regulation* period (1950–1980s). Although humans had been building some flood defences along the Ebro River for centuries, it was not until the highest flood of the century, which occurred in 1961 (instantaneous peak of $4,130 \text{ m}^3 \text{ s}^{-1}$ at the gauging station of Zaragoza, 74-year recurrence interval), that the number of dykes, levees, ripraps, breakwaters, concrete walls and dredging operations dramatically increased in the Middle Ebro (from ~ 5 defence works year^{-1} before 1961 to ~ 25 defence works year^{-1} in the period 1961–1980) (Ollero 2007). This fact severely limited channel migration. In parallel, the largest reservoirs currently operating upstream of the study area were built at the beginning of this period (Fig. 1; Table 1). Surprisingly, these new dams did not alter significantly the timing (Fig. 2a) and flood pulses (Fig. 2b;

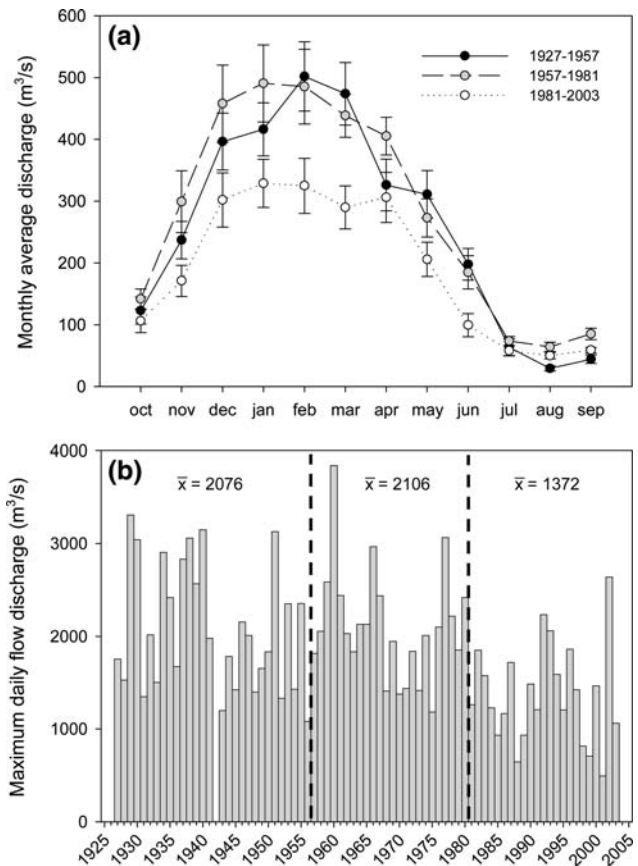


Fig. 2 **a** Monthly mean flow discharge and **b** annual maximum daily flow discharge at the gauging station of Zaragoza ($41^{\circ}39'N$, $0^{\circ}52'W$) (Middle Ebro River, NE Spain) for the pre-regulation (1927–1957), regulation (1957–1981) and stabilization period (1981–2003)

Table 2) of the flow regime with regard to the previous period, although it is likely that the peak in precipitation registered in almost all the Ebro River Basin during the 1970s (Abaurrea and others 2002) dampened the effects of dams on the hydroperiod (Cabezas and others 2009). Likewise, the dams probably limited the provision of sediments as well and, together with river embankment, they may have increased river bed incision, and

Table 1 Largest dams currently operating in the Ebro River Basin upstream of the study area (Natural Reserve of the Alfranca Galachos, 41°36'N, 0°46'W)

	Capacity (hm ³)	River	Distance from study area (km)	Year of completion
Ebro Dam	540	Ebro	350	1952
Yesa Dam	447	Aragón	115	1959
Sotonera Dam	189	Sotón-Gállego	65	1963
Tranquera Dam	84	Piedra	165	1959
Mansilla Dam	68	Najerilla	190	1960

Table 2 Bankfull discharge, flood frequency and duration at the gauging station of Zaragoza (41°39'N, 0°52'W) (Middle Ebro River, NE Spain) for the pre-regulation (1927–1957), regulation (1957–1981) and stabilization (1981–2003) periods; a flood event is defined as one exceeding 600 m³ s⁻¹

	1927–1957	1957–1981	1981–2003
Bankfull discharge (m ³ s ⁻¹)	1980	1957	1410
Flood frequency (events year ⁻¹)	7.6	6.6	4.2
Flood duration (days event ⁻¹)	5.1	5.9	4.4

progressively fixed the channel (Ollero 2007). In addition, land reclamation by agriculture continued with the expansion of irrigation, reducing the surface occupied by natural vegetation (Cabezas and others 2009).

(3) *Stabilization* period (1980s–present). The construction of flood protection structures in the Middle Ebro was intensified in the 1980s (up to 32 works year⁻¹) (Ollero 2007), covering the meander concave bars of the entire channel (e.g. study area, Fig. 1). The mean annual discharge in the Ebro since the 1980s has decreased by 30% with regard to the previous two periods due to, among other factors, a shift to high water-demand crops, such as rice, fruit and vegetable (Frutos and others 2004; Cabezas and others 2009). In addition, the magnitude, frequency, duration, and variability of flood events were also reduced (Fig. 2; Table 2), while the frequency and duration of low-water events increased compared to the previous period (Ollero 2007). These low-water events were mainly recorded during the summer but they also appeared in spring and autumn (Frutos and others 2004). In fact, June's mean flow discharge has notably decreased in this period (Fig. 2a). Finally, agricultural land cover has been maintained until the present time, limiting the surface occupied by natural forests to only 4.5% of the 5-km mean width Middle Ebro floodplain (Ollero 2007).

Forest Patches Age-Classification and Plot Selection

Among existing aerial missions, the photos of 1927, 1957, 1981 and 2003 were chosen as representative of the three above-mentioned periods, and used to classify forest patches according to their age. Each image was rectified and

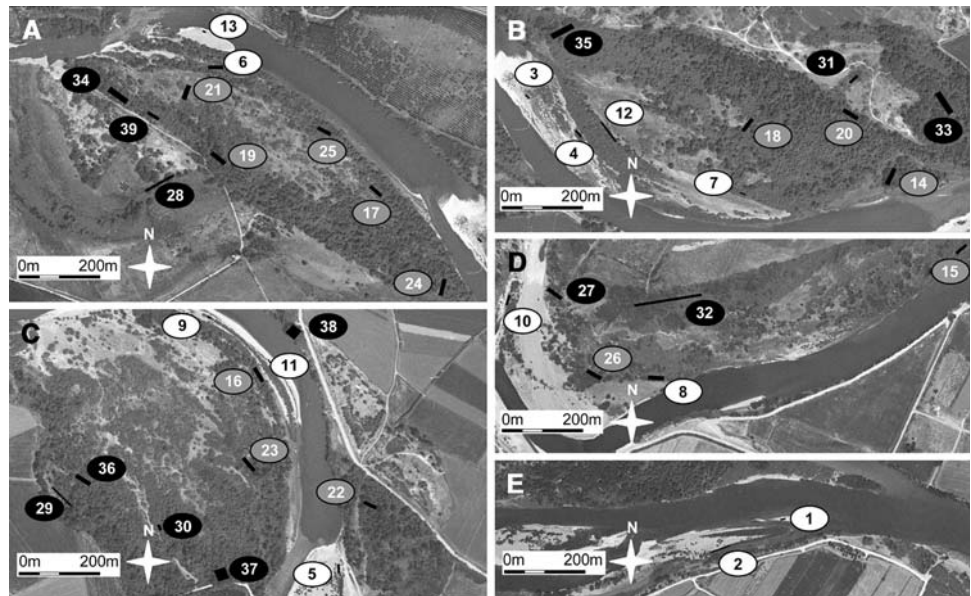
georeferenced and the open water, barren sites (potential recruitment sites), non-woody vegetation (herbaceous and macrophytes), natural floodplain forests, and anthropic landcovers (including poplar crops and agricultural fields) were digitized using ArcGis 9.2[®]. Patches were identified through a fixed scale of 1:3,000 following patterns of texture, colour, vertical structure, position in the landscape and previous channel migration dynamics. A stereoscope was used to improve picture interpretation. The digitation was limited to the area flooded during the 10-year flood of February 2003 (2,230 ha) (map provided by the Ebro River Basin Administration; Losada and others 2004).

The four maps compiled from the aerial photos were used to distinguish the following three age-forest categories on the aerial photo of 2003: *young* forests (i.e., an absence of forest in 1981's image but a presence in 2003's) (<25 years old), *mature* forests (i.e., an absence of forest in 1957's image but a presence in 1981's) (25–50 years old) and *old* forests (i.e., the forest was present before 1957) (>50 years old). For a detailed field survey, a total of 13 patches of each category (i.e., 39 patches in total) were selected (Fig. 3). The following criteria were used to select a patch: (1) evidence of the establishment from an initial cohort of pioneer trees; (2) patch size ≥0.25 ha; (3) location within a distinctive geomorphic landform (gravel bar, natural levee of permanent/intermittent channel, floodplain terrace); (4) no evidence of current or former human disturbance, such as planting, grazing, cutting, irrigation, gravel pit extraction or cultivation and (5) all the successional pathways and the full range of elevations and distances from the main channel had to be represented.

The same procedure was applied to the aerial photographs of 1957 and 1981 to calculate the area occupied by *pioneer* forests (i.e., <25–30 years old, presence of forest in the image under examination but absence in the previous image), *non-pioneer* forests (i.e., >25–30 years old, presence of forest in the image under examination and in the previous image) and *undated* (presence of forest in the image under examination but absence of previous image) for each year.

Only one field plot was randomly placed in each patch to avoid pseudoreplication bias and to cover a broader area. Rectangular plots were used because they adapted geometrically to the usually elongated riparian forest patches.

Fig. 3 Localization of the study sites within the Natural Reserve of the Alfranca Galachos, Middle Ebro River (NE Spain). Rectangular sampled plots are shown in black. White, grey and black labels indicate *young* (colonized after 1981 and numbered from 1 to 13), *mature* (colonized after 1957 and numbered from 14 to 26) and *old forest* plots (colonized before 1957 and numbered from 27 to 39), respectively. The location of the five aerial images is represented in Fig. 1 and denoted by uppercase letters. The aerial photo was taken in 2003



Plot dimensions varied proportionally to tree height, to capture the maximum variability without over-sampling (van Pelt and others 2006). The long axis of each plot was made at least twice the dominant height (i.e., mean height of the two highest trees). The short axis was long enough to make the plot area = dominant height². In all, plot areas ranged from 25 to 1,000 m².

Tree Survey

Forest structure and health were analyzed within each field plot. Every stem in the plot was registered for species, diameter and health status from late-spring to mid-summer of 2006 and 2007.

Three size classes were defined for each species, according to their stem height and diameter: seedlings (0.3 m ≤ height < 1.3 m or diameter at breast height [d.b.h., 1.3 m] < 2.5 cm), saplings (height ≥ 1.3 m and 2.5 ≥ d.b.h. < 7.5 cm) and adults (height ≥ 1.3 m and d.b.h. ≥ 7.5 cm). An exception had to be made with saltcedars because of their shrubby architecture: seedlings (0.3 m ≤ length < 1.3 m or diameter at 0.30 m [d30] < 2.5 cm), saplings (length ≥ 1.3 m and 2.5 ≥ d30 < 7.5 cm) and adults (length ≥ 1.3 m and d30 ≥ 7.5 cm). In addition, saltcedars could not be described at species level in the absence of inflorescence during the sampling period. Adult stems were further arbitrarily divided into small (d.b.h.: < 22.5 cm) and large (d.b.h.: ≥ 22.5 cm). Stem density (stems ha⁻¹) was calculated including all size classes. Within an age-forest category, the presence (%) of each species was calculated as the percentage of plots where at least one stem of the given species was present.

Regarding health status, every stem was registered as dead or alive. Additionally, the canopy dieback (i.e., a

measure of the extent of leafless branches in the tree canopy) of each adult live stem was estimated visually by class defined according to the following criteria: none = foliage remaining rich and dark green; moderate = existing dead or precociously senescing branches (excluding self-pruning and infections by pathogens as causes of death); severe = senescing stem.

Statistical Analyses

As most of the data did not follow a normal distribution, non-parametric statistics were used. The non-parametric Kruskal–Wallis *H* test, analogous to the one-way analysis of variance (ANOVA), followed by Mann–Whitney *U* pairwise comparisons, were used to test for significant differences between species or size classes for the measured variables. A particular non-parametric test analogous to the one-way ANOVA (the Jonckheere–Terpstra test), which assumes that there is an *a priori* ordering (ascending or descending) in the chronosequence, was used to analyze whether the differences found in the measured variables followed age order. SPSS 13.0 was used for all statistical analyses.

Results

Changes at Landscape Scale

The total surface occupied by natural forests and barren sites (potential recruitment sites) in the 10-year floodplain decreased progressively from 1927 to 1981 (Fig. 4; Table 3). During the *stabilization* period (1981–2003), the loss of barren sites continued, but at a lower rate and the

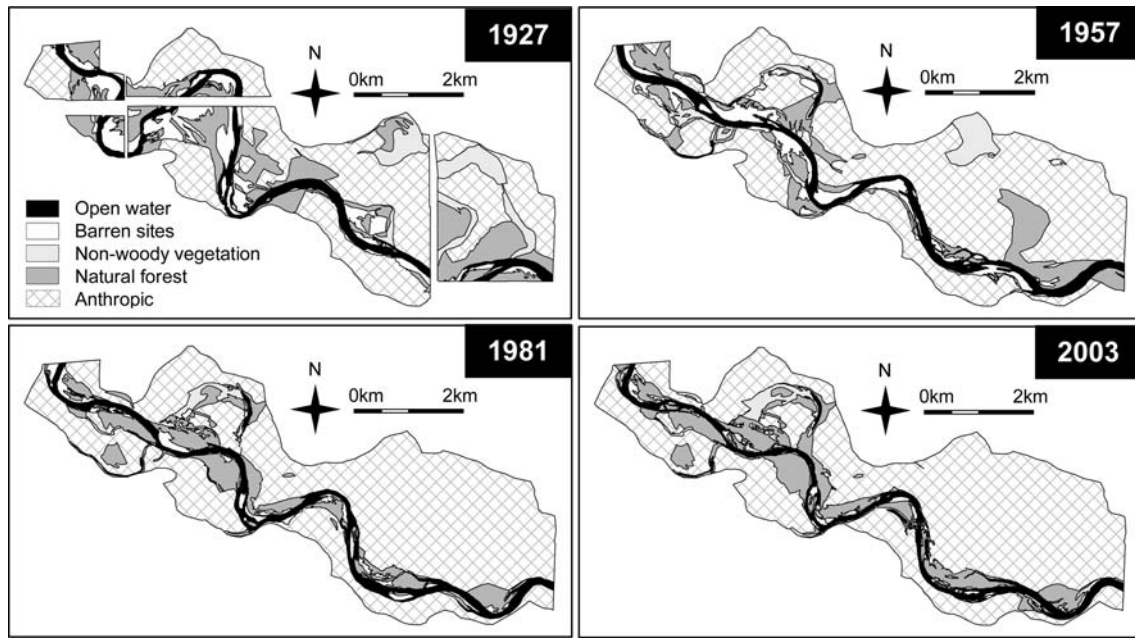


Fig. 4 Historical evolution of the 10-year recurrence interval floodplain in 2003 (2,230 ha) during the twentieth century in the Natural Reserve of the Alfranca Galachos, Middle Ebro River (NE Spain)

Table 3 Floodplain area (ha) occupied by the different ecotopes within the 10-year recurrence interval floodplain in 2003

	Floodplain area (ha)			
	1927	1957	1981	2003
Open water	194	170	171	135
Barren sites	254	167	77	43
Non-woody vegetation	157	114	40	65
Anthropic	1,125	1,441	1,654	1,627
Natural forests	500 (100%)	338 (100%)	288 (100%)	360 (100%)
Pioneer	0	174 (52%)	188 (65%)	110 (31%) ^a
Non-pioneer	0	126 (37%)	100 (35%)	250 (69%) ^b
Undated	500 (100%)	38 (11%)	0	0
Total	2,230	2,230	2,230	2,230

^a Young forests

^b Mature + old forests

floodplain forest recovered to such an extent that the total floodplain forest area in 2003 was higher than in 1957. However, within the natural forest category, the area occupied by *pioneer* forests decreased in the period 1981–2003 (Table 3). As a result, the natural forests shifted from a dominance of *pioneer* units in 1957 and 1981 to a dominance of *non-pioneer* forest in 2003.

Stem Density

The live stem density (excluding dead stems) along the chronosequence (Fig. 5) was highly contrasted, averaging 24,804 stems ha⁻¹ in the *young*, 2,444 stems ha⁻¹ in the *mature* and 3,843 stems ha⁻¹ in the *old* plots ($P = 0.001$, d.f. = 2, $X^2 = 13.931$, Kruskal–Wallis test). In the *young*

forests, the high stem density was mainly due to the contribution of *Tamarix* spp., *P. nigra* and, to a lesser extent, *S. alba*. *Tamarix* spp. also exhibited the highest live stem densities within *mature* forests, but with a mean density almost twentyfold lower than in the *young* forests, followed by *F. angustifolia* and *U. minor*. In the *old* forests: *U. minor*, *Tamarix* spp. and *P. alba*, which surprisingly was significantly present only in this age category, contributed the most to live stem density.

Mortality

The proportion of dead stems (Fig. 5) was lower in *young* (12%) than in *mature* (23%) and *old* plots (27%) ($P = 0.022$, $Z = -2.284$, Mann–Whitney $U = 40$ and

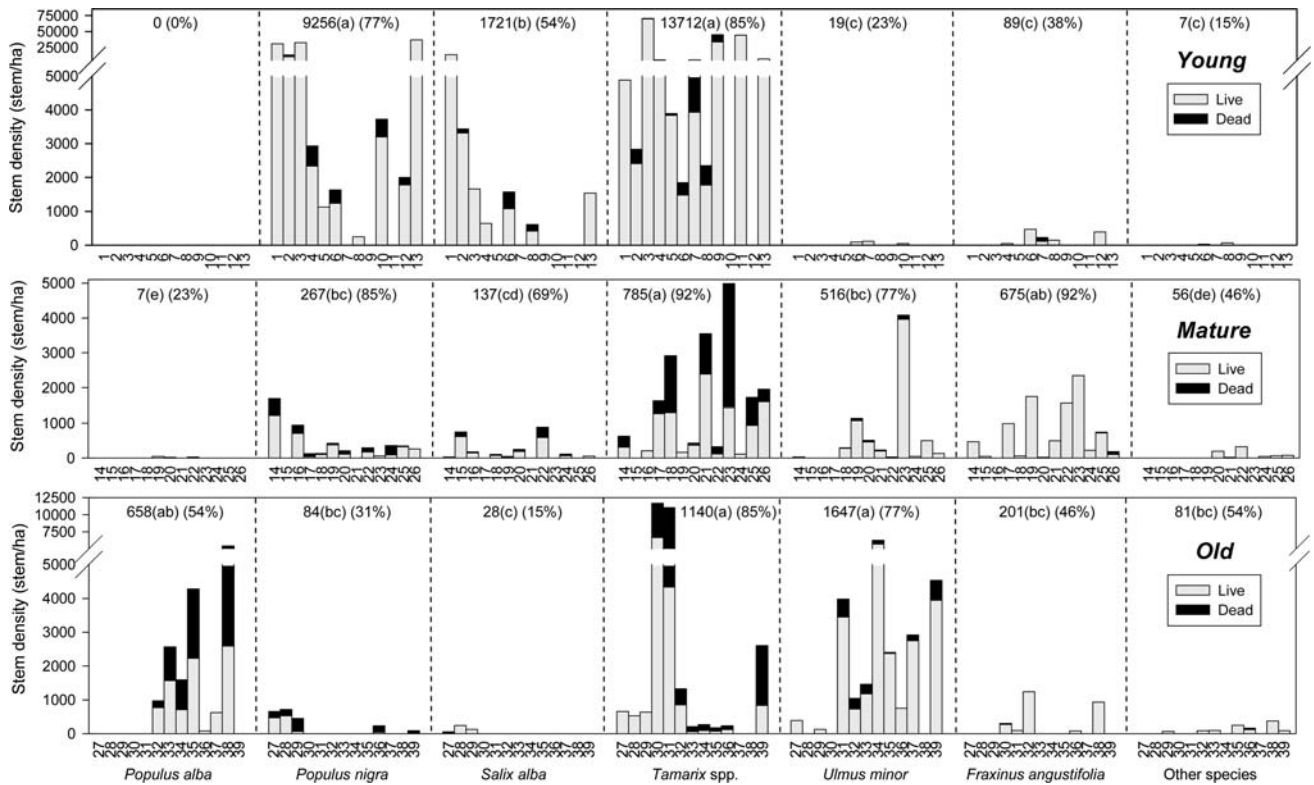


Fig. 5 Stem density along the chronosequence with *young* (tree colonization after 1981), *mature* (tree colonization after 1957) and *old* (tree colonization before 1957) forest types. For each tree species: the bars represent stem density in each plot (denoted in the X axis), numbers indicate the mean live stem density ($n = 13$); percentage (in

brackets) indicate species presence, calculated as the proportion of plots where at least one stem of the species was present. For each forest type: different letters indicate significant differences among species in live stem density (Mann–Whitney pair wise comparisons)

$P = 0.007$, $Z = -2.643$, Mann–Whitney $U = 33$, respectively). Indeed, the proportion of standing dead stems increased as plot aged ($P = 0.004$, Std. Jonckheere-Terpstra Statistic = 2.847). In *young* forests, mortality was equally distributed among the species. However, in *mature* and *old* forests, mortality was generally higher in *P. alba*, *P. nigra*, *S. alba* and *Tamarix* spp. than in *F. angustifolia*, *U. minor* and other species ($P < 0.05$, Mann–Whitney tests) (Fig. 5). Mortality was regularly distributed among the size classes within each species. The only exception to that trend was *P. alba*, whose proportion of dead small stems (62%) was higher than that of dead adults (11%) ($P = 0.031$, $Z = -2.155$, Mann–Whitney $U = 6.0$).

Size Class Distribution

The distribution of live stems into size classes (Fig. 6) provided complementary information to stem density and a better understanding of the population structure and dynamics of each species in the chronosequence. As expected, small stems (i.e., seedlings and saplings) clearly predominated in the *young* forests, with a mean live stem density (excluding dead stems) of 23,329 stems ha^{-1} ,

compared to adults with 1,474 stems ha^{-1} ($P = 0.013$, $Z = -2.491$, Mann–Whitney $U = 36.0$). *Tamarix* spp., *P. nigra* and *S. alba* shared a similar population structure and were in a very active recruitment phase. In *mature* and *old* plots, the live small stems were dominant within *F. angustifolia* and *U. minor* species ($P = 0.010$, $Z = -2.562$, Mann–Whitney $U = 37.5$ and $P = 0.014$, $Z = -2.451$, Mann–Whitney $U = 83.0$, respectively). However, the density of live small stems of *P. nigra* and *S. alba* was extremely low, and significantly higher in the rest of species ($P < 0.05$, Mann–Whitney tests), although the small stems densities of *P. alba*, *Tamarix* spp., *U. minor* and *F. angustifolia* never reached the ones found in *young* forests ($P < 0.05$, Mann–Whitney statistics). Furthermore, the largest individuals found in the study area were *Salicaceae* spp., whereas species in the remaining families rarely attained sizes larger than 22.5 cm in d.b.h.

Canopy Dieback

The likelihood of finding an individual affected by canopy dieback (Fig. 7) was high in *mature* (74%) and *old* forests (52%) and significantly lower in *young* forests (21%)

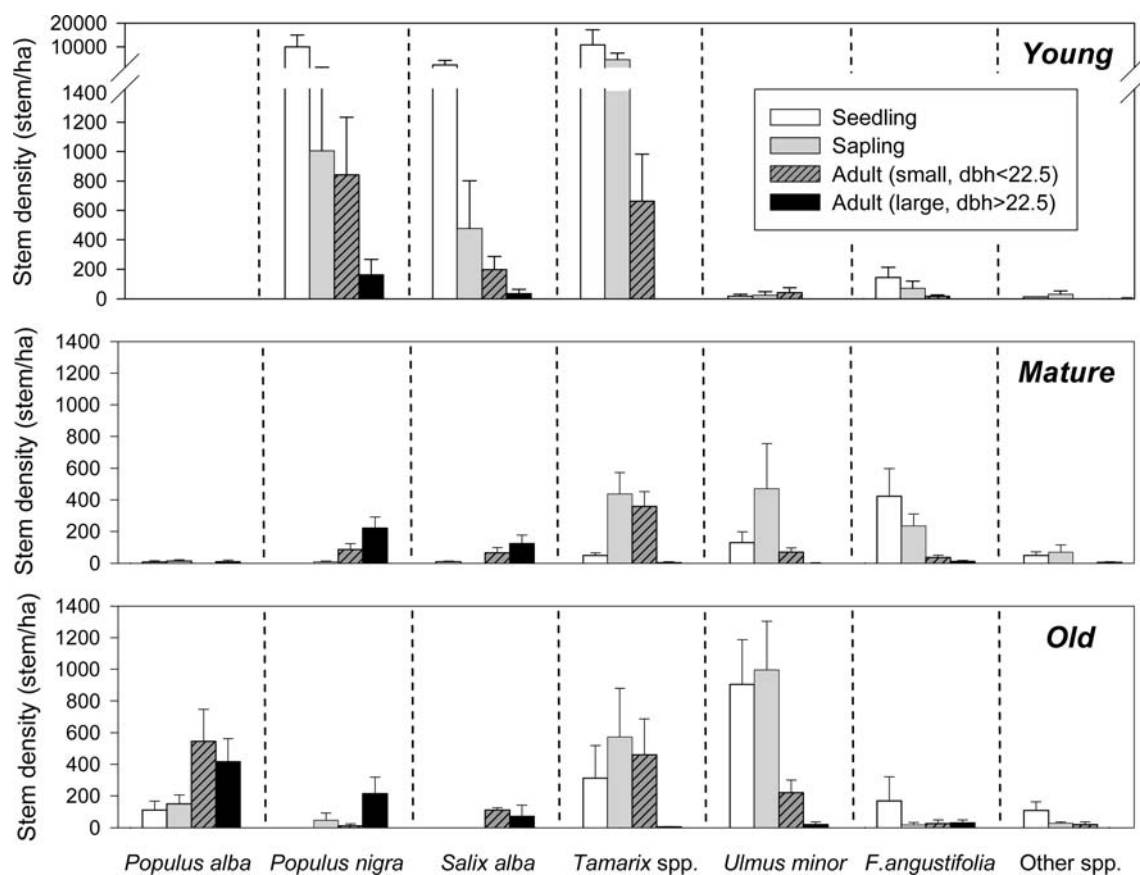


Fig. 6 Size class distribution along the chronosequence with *young*, *mature* and *old* forest types. For each species, the bars represent the

mean live stem density of each size class in the plots where the species was present. The error bars represent ± 1 standard error of the mean

($P < 0.001$, $Z = -3.583$, Mann–Whitney $U = 5.0$ and $P = 0.025$, $Z = -2.250$, Mann–Whitney $U = 25.0$, respectively). It was also very variable between plots within the same age-category. In *young* forests, no significant differences were found among species in the percentages of live adult stems exhibiting dieback. Differences appeared in both *mature* and *old* plots ($P < 0.001$, d.f. = 6, $X^2 = 37.391$ and $P = 0.032$, d.f. = 6, $X^2 = 13.771$, respective Kruskal–Wallis tests). In particular, almost all (>90%) *P. nigra* and *S. alba* were affected by dieback, but the latter more severely than the former. Although *Tamarix* spp. were less affected than the other two species (*P. nigra*: $P = 0.018$, $Z = -2.375$, Mann–Whitney $U = 91.0$ and *S. alba*: $P = 0.001$, $Z = -3.323$, Mann–Whitney $U = 37.5$, grouping *mature* and *old* plots), >60% of the population still exhibited canopy dieback. On the other hand, *P. alba* were much less affected than their family counterparts (*P. nigra*: $P = 0.001$, $Z = -3.372$, Mann–Whitney $U = 7$ and *S. alba*: $P < 0.001$, $Z = -3.727$, Mann–Whitney $U = 0$; grouping *mature* and *old* plots). The rest of the species rarely exhibited dieback, although the majority of elms were affected by the *Dutch Elm Disease* (but still not causing branch dieback).

Discussion

Forest Structure

P. nigra and *S. alba* are the most widespread *Salicaceae* species in the European floodplains, exhibiting clear pioneer establishment traits (van Splunder and others 1995; Barsoum and Hughes 1998; Hughes and others 2001; Guilloy-Froget and others 2002). Their widespread presence and high dominance (especially as live small-size stems) in the *young* forests reflect a relatively recent colonization and indicate that the hydrogeomorphic requirements for their establishment were met during the *stabilization* period, but only in 5% of the 10-year floodplain. In contrast with *young* forests, *P. nigra* and *S. alba* exhibited low small-stem densities in *mature* and *old* forests, which is typical of a community with low recruitment levels and provides evidence of an unsuitable hydrogeomorphic regime at site scale to recruit individuals of the two species in the remaining 11% of the 10-year floodplain covered by natural forests.

An unusual feature of the Ebro River, compared with the Northern-European environments, is not only the presence of *Tamaricaceae* spp. but also their high density

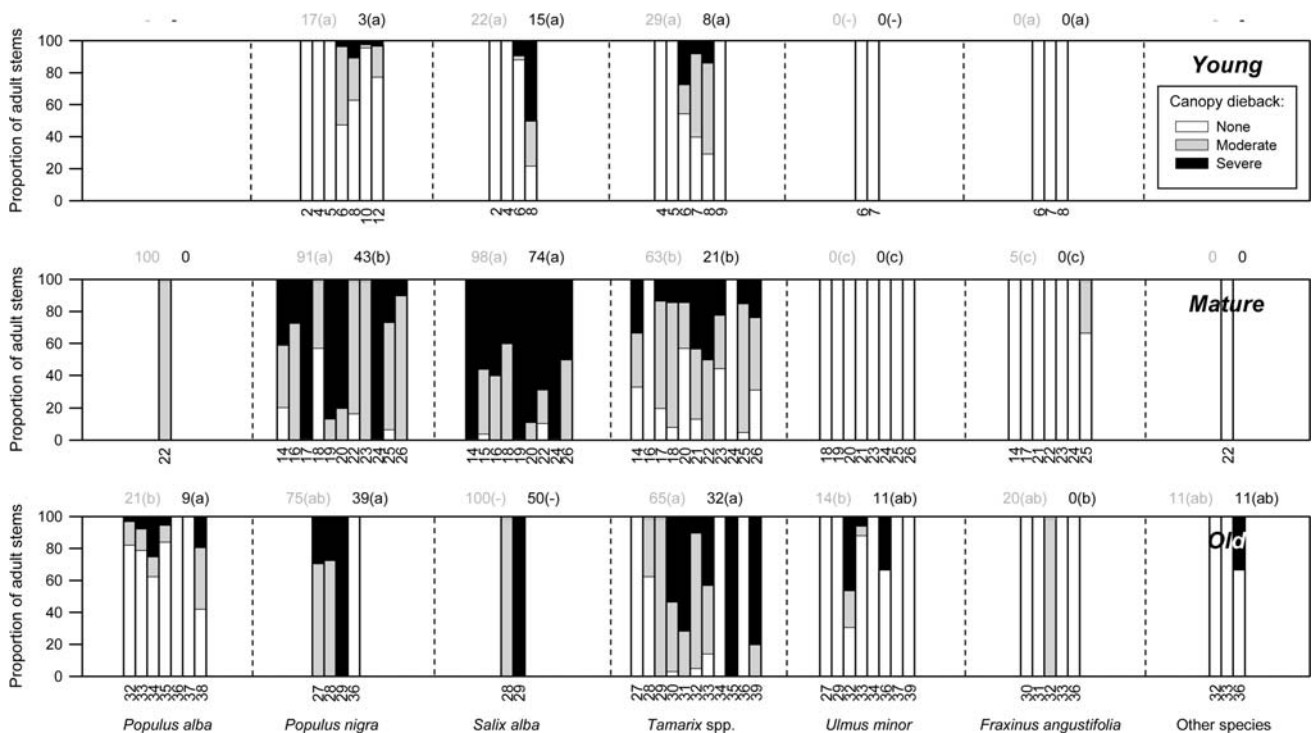


Fig. 7 Canopy dieback along the chronosequence with *young*, *mature* and *old* forest types. For each species: bars represent the 100% of live adults in each plot (denoted in X axis); numbers represent the mean proportion of live adults exhibiting canopy dieback (grey) and severe canopy dieback (black). For each forest type and species:

different letters indicate significant differences among species in the proportion of live adults exhibiting canopy dieback (grey) and severe canopy dieback (black) (Mann–Whitney pair wise comparisons). Analyses were not performed when a given species was present in less than three plots

and dominance. Like *P. nigra* and *S. alba*, their high density of small stems suggests that their hydrogeomorphic requirements were met during the *stabilization* period. *Tamarix* spp. are also prolific seed producers which need recently disturbed areas for seedling establishment (di Tomaso 1998). However, it is likely that they adapted better than *Salicaceae* to the changing hydrogeomorphic conditions in the Middle Ebro, as they could maintain a higher stem density than *P. nigra* and *S. alba* along the chronosequence. Traditionally, *Tamaricaceae* spp. have been considered to respond to the impacts derived from river regulation better than *Salicaceae*, principally because of their greater tolerance to water stress and salinity (Glenn and Nagler 2005) and their longer and later dispersal of seeds (Gladwin and Roelle 1998; Roelle and Gladwin 1999; Roelle and others 2001). However, the high stem densities of saltcedar in *mature* and *old* forests may be due not to a more effective regeneration than *P. nigra* and *S. alba*, but to their shrubby architecture. Indeed, Cleverly and others (1997) reported lower stem growth rates (narrower annual tree rings) in *Tamarix* compared to co-existing *Salix*. This would partly explain why saltcedars rarely attained diameters larger than 22.5 cm (Fig. 6).

P. alba seems the species most affected by regulation, since it only appeared in the stands colonized before the *regulation* period (i.e. *old* stands). The few individuals detected in *mature* forests (plots 19, 20 and 22) could be root suckers from adjacent *old P. alba* stands. Although not specifically addressed in this study, the origin of most *P. alba* stems is probably vegetative, as homogeneous phenology and morphology has been observed in *P. alba* stands (i.e., stems grouped by sex, neighbour females coupling flowering and seed dispersal periods) (González, *personal observation*). Legionnet and others (1997), Gom and Rood (1999) and Barsoum and others (2004) described significant vegetative propagation in some *Populus* species. However, to our knowledge, there is no study of *P. alba* vegetative regeneration in the existing literature. The apparent sexual regeneration failure of *P. alba* during the last few decades is especially surprising, as it has been recently shown that their sexual regeneration strategies are similar to those of other riparian *Populus* (González and others 2010), and deserves further research.

The loss of riparian habitats, especially those in the outer floodplain, caused by the increase of the area occupied by crops, together with the spreading of the *Dutch Elm Disease*, have caused the disappearance of most *U. minor*

stands in Spain (García Nieto and Morla 1990; Lara and others 2004; López-Almansa 2004). However, the high sprouting rate from surviving roots of unhealthy trees, as reported for Spanish elms (López-Almansa 2004), could explain the widespread presence and relative abundance observed in the Middle Ebro. *Ulmus* high regeneration, and high mortality in young trees caused by *Dutch Elm Disease* were also reported in the Upper Rhine nearby Strasbourg (France) by Trémolières and others (1998). *F. angustifolia* is frequently found in most river margins of the Northern Iberian Peninsula, including the Ebro River, but rarely dominating (Lara and others 2004), as happens in this study area. Oszlányi (1997) also described a vigorous *F. angustifolia* regeneration (up to 240,000 small stems ha⁻¹) under a declining tree layer of *P. nigra*, *S. alba* and *P. alba* in a disconnected forest in the Slovakian Danube River. Both *U. minor* and *F. angustifolia* establish and regenerate more effectively in *mature* and *old* forests, which suggests not only that they are less opportunistic than the other species, but also that their regeneration is not so dependent on recurrent fluvio-geomorphic events. Nevertheless, it is yet unknown whether succession under the current hydrogeomorphic regime would lead to a more widespread presence of larger size classes of these hardwood species in the future.

In short, *P. nigra*, *S. alba* and *Tamarix* spp. may be defined as pioneer tree species in the Ebro River, whereas *P. alba*, *U. minor* and *F. angustifolia* would rather be considered as non-pioneer species.

Forest Health

The higher canopy dieback of pioneer species observed in *mature* and *old* stands might be due not only to natural senescence but also to the hydrologic regime change, as most of the trees colonized those patches before 1981 and, therefore, under different hydrological conditions. Thus, human activity might be causing a decline of *P. nigra* and *S. alba* along the Ebro River, being both species extremely affected by dieback, as it has been reported for other *Salicaceae* species along dam-regulated rivers elsewhere (Rood and Heinze-Milne 1989; Rood and Mahoney 1990; Johnson 1992; Rood and others 1995; Stromberg and others 1996; Cordes and others 1997; Merritt and Cooper 2000; Muller and others 2002; Williams and Cooper 2005). The better health status of saltcedars in the Middle Ebro compared to the *Salicaceae* species is consistent with their abovementioned properties, although saltcedars could also be sacrificing branches in response to water stress. The rest of the species are much less affected by dieback, as found by Oszlányi (1997), who also reported partial destruction of the canopy of almost all the *P. nigra* and *S. alba* individuals in the Slovakian Danube River, whereas the co-occurring *F. angustifolia*

showed an excellent health status. Nevertheless, the great variability in dieback among plots within the same age category also suggests the need for more detailed studies aiming to identify the ultimate factors driving this morphological response to hydric stress.

Likewise, the change in hydrologic conditions could have increased mortality among pioneers in *mature* and *old* forests. However, the higher mortality among small *P. alba* stems could be due to natural thinning in the absence of light under the usually dense and healthy canopy of *P. alba*, as young individuals of *Salicaceae* do not tolerate shade (Sacchi and Price 1992; Cooper and others 1999).

Management Implications

The area occupied by *non-pioneer* forests relative to *pioneer* forests today is much higher than it was in the absence of regulation (Table 3). If the current hydrogeomorphic patterns are maintained, it is likely that most of the existing forests will not disappear but will keep ageing (as it occurred during the *stabilization* period, Table 3), whereas new *pioneer* forests will be confined to smaller and more dynamic areas, where the effect of floods is recurrently destructive (usually close to the main channel or in-channel areas; Figs. 3, 4). Consequently, stands with structural characteristics like the ones observed in *young* forests would be progressively less common, while stands structurally similar to *mature* and *old* forests would predominate. Under this scenario, *P. nigra*, *S. alba* and, to a lesser extent, *Tamarix* spp. populations could notably decline in the Middle Ebro floodplain forests during the twenty first century. Their progressive substitution by *U. minor* and *F. angustifolia* is uncertain. In the short and medium term, the maintenance of *P. alba* populations at patch scale seems guaranteed. However, at landscape scale, it is unlikely that *P. alba* will colonize new sites, so its future in the Middle Ebro is also compromised. Nevertheless, as the regulation infrastructures are relatively recent (less than 50 years), their effects on vegetation structure are probably not yet fully manifested (Johnson 1998; Katz and others 2005) and our predictions should be taken with caution. Another uncertainty may come from the exotic species (e.g., *A. negundo*, *R. pseudoacacia*) already detected, which are not yet invasive in the Middle Ebro, and from the effects of climate change. Fortunately, the potential for ecological restoration remains high. Even in pristine environments, regeneration of pioneers can systematically fail for years because the precise hydrologic conditions required for seedling establishment are only occasionally satisfied (Mahoney and Rood 1998). Thus, these populations have developed the faculty of keeping an enormous reproductive potential (Gom and Rood 1999; Karrenberg and others 2002) and, when appropriate restoration measures are taken, successful regeneration occurs rapidly

and effectively (Rood and Mahoney 2000; Stevens and others 2001; Stromberg 2001; Rood and others 2003, 2005).

In this sense, the most effective restoration approach in the Middle Ebro should focus on the recovery of some hydrogeomorphic dynamism (i.e., channel migration, periodic creation of new barren sites, reactivation of secondary channels, meander cut-offs, renaturalized hydroperiod, etc.) both at basin and reach scale, within the current socio-economic context (Comín and others 2005). Thus, the ideal hydrogeomorphic regime would not necessarily be the pre-regulation state but one 'renaturalized', which led to a self-sustainable forest structure at patch and landscape scale, guaranteed their ecological functions and provided services to society (Dufour and Piégay 2009). However, more research is needed to define those ideal conditions locally. At this point, the most urgent actions to be implemented seem those aimed at maintaining a mosaic of forest patches of different age by creating opportunities for the periodic colonization of new *pioneer* forests. These actions should be implemented at basin and reach scale.

At basin scale, water demand by society is expected to increase in the future (Cabezas and others 2009), so the 'renaturalization' of the flow regime through dam operations (e.g., raise of low water levels, release of controlled floods and sediments) is not likely to occur.

At reach scale, returning part of the land back to the river (the 'mobility space' concept) seems more feasible. Such approach has been recognized as the most effective and cheapest strategy to promote the creation of new riparian forests, as it resizes rivers to their new hydrologic situation and allows them to do the work (Middleton 2002; Rood and others 2005; Ollero 2007; Stromberg and others 2007). The implementation of this strategy might include some of the following measures: removal and set back of flood defence structures; recovery of agricultural lands which are no longer profitable; reconversion of poplar crops (~ 50 ha in the 10-year floodplain in 2003) to natural floodplain after harvest; lowering floodplain heights to improve the erosive capacity of floods and in-channel reallocation of extracted sediments to facilitate the creation of nursery sites; locally controlled flooding to promote regeneration of desirable tree species, especially *P. nigra*, *S. alba*, *Tamarix* spp. and *P. alba*, preceded by mechanical disturbance of the substrate and removal of ruderal vegetation in stabilized sites; and prohibition of grazing in potential nursery sites.

In case of economic shortage, these actions should be prioritized over the maintenance of the existing senescent forests (those that increased during the *stabilization* period) although, meanwhile, drops in water table levels and the duration of low-water periods should be controlled to minimize the high canopy dieback and mortality observed in *non-pioneer* forests. A similar forest management

strategy should be applied to other Mediterranean watersheds with hydrogeomorphic constraints similar to those of the Middle Ebro River.

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