Journal of Environmental Management 202 (2017) 392–402

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Research article

Long-term monitoring for conservation management: Lessons from a case study integrating remote sensing and field approaches in floodplain forests

Patricia María Rodríguez-González [a, *], António Albuquerque [a, b], Miguel Martínez-Almarza [c], Ricardo Díaz-Delgado [c]

[a] Centro de Estudos Florestais, Instituto Superior de Agronomia, Universidade de Lisboa, Portugal
[b] Ecofield, Monitorizações, Estudos e Projectos, LDA, Carcavelos, Portugal
[c] Laboratorio de SIG y Teledetección (LAST), Estación Biológica de Doñana, CSIC, Spain

ARTICLE INFO

Article history:
Received 15 May 2016
Received in revised form
21 January 2017
Accepted 27 January 2017
Available online 9 February 2017

Keywords:
Foundation species
Fraxinus
Resilience
Riparian vegetation
Salix
Wetland hydrology

ABSTRACT

Implementing long-term monitoring programs that effectively inform conservation plans is a top priority in environmental management. In floodplain forests, historical pressures interplay with the complex multiscale dynamics of fluvial systems and require integrative approaches to pinpoint drivers for their deterioration and ecosystem services loss. Combining a conceptual framework such as the Driver-Pressure-State-Impact-Response (DPSIR) with the development of valid biological indicators can contribute to the analysis of the driving forces and their effects on the ecosystem in order to formulate coordinated conservation measures. In the present study, we evaluate the initial results of a decade (2004–2014) of floodplain forest monitoring. We adopted the DPSIR framework to summarize the main drivers in land use and environmental change, analyzed the effects on biological indicators of foundation trees and compared the consistency of the main drivers and their effects at two spatial scales. The monitoring program was conducted in one of the largest and best preserved floodplain forests in SW Europe located within Doñana National Park (Spain) which is dominated by *Salix atrocinerea* and *Fraxinus angustifolia*. The program combined field (*in situ*) surveys on a network of permanent plots with several remote sensing sources. The accuracy obtained in spectral classifications allowed shifts in species cover across the whole forest to be detected and assessed. However, remote sensing did not reflect the ecological status of forest populations. The field survey revealed a general decline in *Salix* populations, especially in the first five years of sampling –a factor probably associated with a lag effect from past human impact on the hydrology of the catchment and recent extreme climatic episodes (drought). In spite of much reduced seed regeneration, a resprouting strategy allows long-lived *Salix* individuals to persist in complex spatial dynamics. This suggests the beginning of a recovery resulting from recent coordinated societal responses to control excessive water extraction in the catchment, highlighting the need for continuing long-term monitoring. The DPSIR framework proved useful as a conceptual tool in analyzing the entire environmental system, while both field and remote sensing approaches complemented each other in quantifying indicator trends, improving the monitoring design and informing conservation plans.

1. Introduction

Multiple pressures threaten biodiversity worldwide and alterations in floodplains are expected from the interaction of global change and escalating water use, particularly in water deficit areas (Vörösmarty et al., 2010). In the Mediterranean region, floodplain forests are resource-rich habitats that support a wide range of ecosystem functions and services which extend far beyond the area they occupy (Stella et al., 2013). Floodplain forests provide ecosystem services, such as those related to water quality, microclimate, wildlife habitats, an energy base for the food web and flood

* Corresponding author. Centro de Estudos Florestais – Edificio Azevedo Gomes, Instituto Superior de Agronomia, Tapada da Ajuda, 1349-017, Lisboa, Portugal.
E-mail address: patri@isa.ulisboa.pt (P.M. Rodríguez-González).

http://dx.doi.org/10.1016/j.jenvman.2017.01.067
0301-4797/

mitigation (MEA, 2005). In spite of their importance, floodplain forests have shrunk significantly due to human impacts at the local and the catchment scale. They are therefore a top-priority target for conservation, management and restoration (Stella et al., 2013). In Europe, the assessment and restoration of degraded riparian areas has become mandatory within the Water Framework Directive and the Biodiversity Strategy to 2020 in the context of the Aichi targets (González del Tánago et al., 2012).

Reliable conservation and restoration of floodplain forests require coordinated actions to reduce the pressures that degrade their status but also to assess ecosystem effects through an adaptive management approach that enhances their resilience to human induced changes (White and Stromberg, 2011; Sanders and Kirschbaum, 2015). Long-term monitoring of riparian plant communities is critical in this respect, not only in order to track biological responses to multiscale anthropic and environmental changes, but also to identify early warning signals of prospective changes in ecosystem functions and services (Cardinale et al., 2012; Ström et al., 2012). A long-term floodplain forest monitoring program is a valuable means of assessing drivers and the effects of pressures, where subtle changes would otherwise remain undetected; it provides scientifically valid information and makes projections about future structure and composition. It is therefore an important tool in adaptive management (Sanders and Kirschbaum, 2015).

Implementing an effective long-term monitoring program of floodplain forest changes is challenging and these type of studies are scarce (but see Nguyen et al., 2015; Sanders and Kirschbaum, 2015). Firstly, this is because a long-term monitoring program is highly resource-demanding. In addition, there is a significant complexity inherent to the dynamics of floodplain systems which integrate multiple spatial (catchment, segment, reach) and temporal scales of variation (Dufour and Piégay, 2009); and finally, it is difficult to document the long-term consequences of changes such as land use and climate change and to pinpoint environmental drivers of deterioration (Nguyen et al., 2015). In order to achieve this, establishing an integrative conceptual framework, such as the drivers–pressures–states–impacts–responses (DPSIR) approach (EEA, 2005), might help to draw links between environmental and anthropogenic drivers on biotic elements, so that more effective conservation strategies can be defined. In a long-term monitoring program, this general framework can be used to evaluate the effects of driving forces and their associated pressures on the state of the environment and their impact on ecosystem functions, as well as to evaluate the effects of societal responses taken during the monitoring period (Haberl et al., 2009).

In addition, the proper choice of reliable, well validated biological indicators (Gumiero et al., 2015) in the design and improvement of monitoring programs is a top-priority goal in conservation management if we are to adequately assess the impact of environment changes on species and subsequently respond with adaptive actions (Haberl et al., 2009). Foundation species (Dayton, 1972) provide a suitable opportunity for such an approach due to their tight relationship with the habitat and their influence on other species in the ecosystem (Corenblit et al., 2011). Foundation riparian trees, such as Salicaceae, interact with environmental change by modulating fluvial system processes (flow velocity, sediment deposition) and consequently affect the physical habitat of other species (Karrenberg et al., 2002). As such, different foundation tree canopies often correspond to different understory communities. Monitoring them is highly worthwhile in order to be able to track alterations liable to have an effect at the community and ecosystem level (Ellison et al., 2005). The use of foundation species as an indicator provides an insight into key ecosystem processes, and allows for management based on information obtained by monitoring these processes.

This study documents the first decade of results from a long term monitoring program in the floodplain forests of Doñana National Park. The main goals of the present study are: 1) to summarize the main drivers and pressures affecting the Doñana floodplain forests within a DPSIR framework; 2) to set up a baseline trend of the dominance, abundance and health (i.e. balanced population structure) of foundation trees to use in the analysis of the environment impact of water availability and pressures from catchment land uses; 3) to test the consistency of trends in foundation trees across indicators analyzed through two monitoring approaches based on remote sensing and field sampling, respectively; and, 4) to assess the utility of the monitoring program for conservation management within a DPSIR framework. We hypothesized that 1) the recent historical (last 20 years) withdrawals of superficial and phreatic water in the La Rocina stream catchment have led to a quantifiable compositional and structural change in foundation tree indicators given the different water requirements of the dominant species and 2) the changes registered at a local scale with field indicators should correspond with dominance shifts in foundation species at the landscape scale by remote sensing imagery analysis.

2. Materials and methods

2.1. Study site description

The Doñana marshes are one of the largest protected wetlands in Europe with an area of 340 km^2, of which 300 km^2 are included in Doñana National Park (Morris et al., 2013; Díaz-Delgado et al., 2010; García Novo et al., 2006). With a typical Mediterranean climate, the flooding cycle starts in September and usually reaches peak levels during the end of boreal winter subject to rainfall variability. The dominant clay and silt substrates of the Doñana marshes are soaked with the first showers and a shallow water layer spreads over the floodable area (Fig. 1). The tributary river network streams into the marshes maintaining water levels as runoff flows in the catchments. Mediterranean water courses vary dramatically during the cycle either causing fast and intense floods or remaining completely dry for long periods.

La Rocina feeds the Doñana marshes from the west collecting runoff and groundwater across 400 km^2 of a sandy catchment. Average annual water discharge from La Rocina to the marshes is about 40 hm^3 (Manzano et al., 2005). While marshes are included in the Doñana National Park, La Rocina and a surrounding 500 m buffer have a minor protection status, limited to the downstream 12 km of its total 19 km length. The floodplain forest canopy is mainly dominated by *Fraxinus angustifolia* Vahl. and *Salix atrocinerea* Brot., with open flooded areas being more abundant as the stream approaches the marshes. Within the forest, each species is generally dominant at opposite positions along a gradient of hydroperiod and flooding frequency: *Salix* is dominant in more flooded areas and *Fraxinus* in the transition to upland vegetation. This stand represents the largest area of a well-preserved Ibero-Atlantic floodplain forest with a Mediterranean climate (Rodríguez-González et al., 2008). Several endemic and threatened species are found in the La Rocina wetland forest. These include *Frangula alnus* Mill. subsp. *baetica* (Willk. & É. Rev.) Rivas Goday, *Rorippa valdesbermejoi* (Castrov.) Mart.-Laborde Castrov, *Utricularia exoleta* R. Br., *Mycropyropsis tuberosa* Romero Zarco y Cabezudo, and *Carex elata* subsp. *tartessiana* Luceño & Aedo.

Following the establishment of the Doñana National Park in 1969, intensive agriculture progressively developed in the surrounding countryside (Haberl et al., 2009) with intensive farming of greenhouse crops irrigated with water pumped from Doñana's

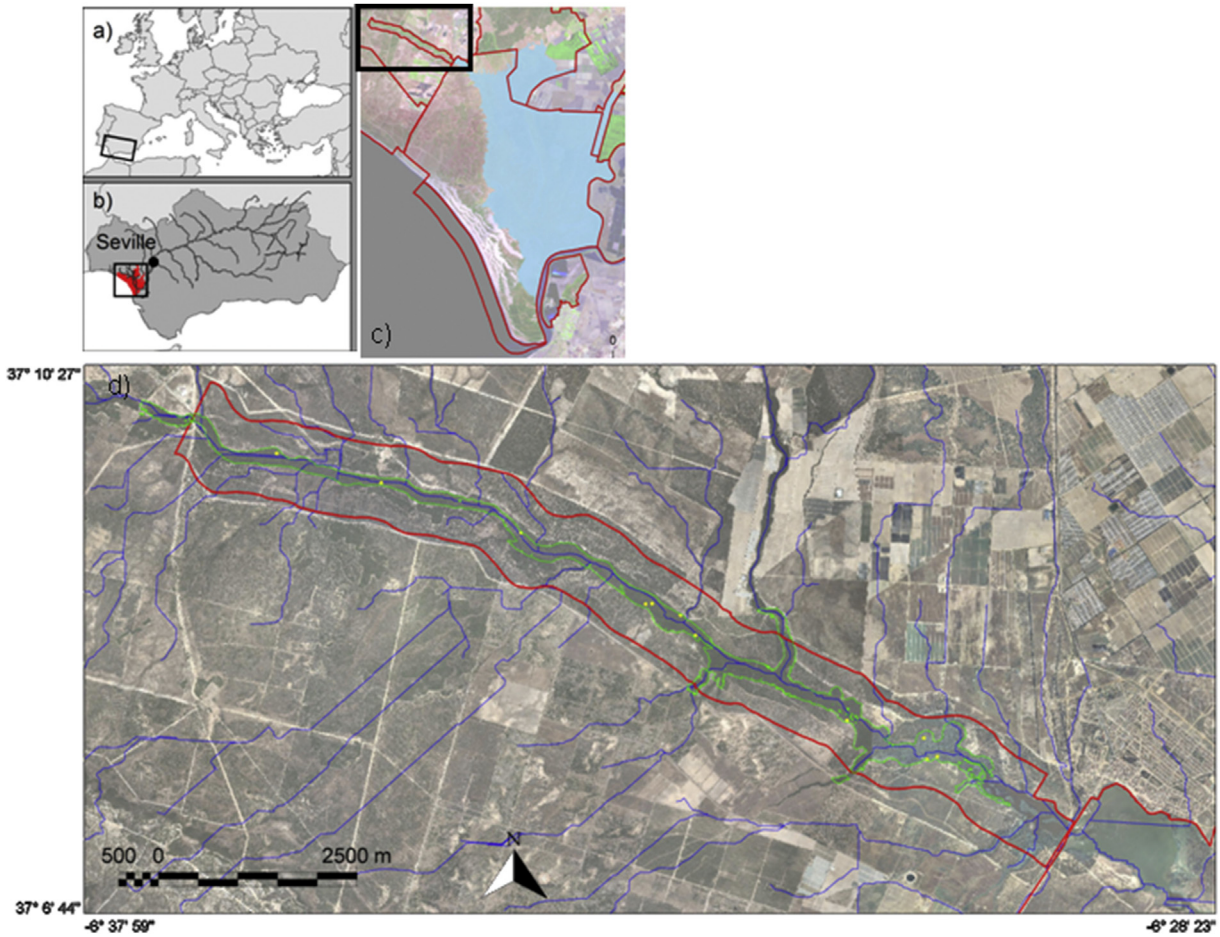

Fig. 1. Location of the study area in the European (a) and Spanish (b) context. Fig. 1 c) also shows the position of the La Rocina stream in relation to the Doñana marshes. Fig. 1 d) details the La Rocina catchment showing the protection limits (red line), the floodplain forest (green line), the hydrographic network (blue line) and the location of permanent plots (yellow dots). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

large aquifer system (Almonte-Marismas). La Rocina catchment has been one of the areas most affected by strawberry farming given that uncontrolled wells are pumping underground water, even in illegally cultivated public lands (Aldaya et al., 2010). During the 1990s, different hydrogeological studies (Custodio et al., 2005, 2009; Manzano et al., 2005) provided evidence of a generalized drop of the piezometric records (water table) in the area (Supplemental Fig. S3). During the last 10 years, the competent administration, *Confederación Hidrográfica del Guadalquivir*, has embarked on a lengthy process of inventorying and filing legal actions. These actions are gradually helping to shut these illegal extractions down. In spite of these efforts, the latest WWF report, integrating research on the state of water in Doñana, points out that the aquifer is "at risk" (WWF España, 2016). On the other hand, large extensions of the most upstream catchment were devoted to eucalyptus and pine afforestations since the 1950s. Water consumption by eucalyptus in the region has been estimated at about 14,000 m^3 ha/year (Aldaya et al., 2010). At the beginning of the 21st century, different management plans implemented clearcutting targeting eucalyptus plantations. From 2005 to 2011 alone, up to 9 km^2 were cleared, representing 2.25% of the total La Rocina catchment (land use cover change analysis based on the 2005 and 2011 SIOSE maps (Spanish Land Use Information System) (Moreira et al., 2008).

In 2002, the Doñana Biological Station together with the National Park Management Board set up an integrated long-term monitoring program with the aim of building the baseline for trend analysis and adaptive decision-making (Díaz-Delgado, 2010). Monitoring features focused on species, habitats and processes, including the impact of anthropogenic drivers. The approach integrates data captured *in situ* by standardized monitoring protocols and information derived from remote sensing image analysis collected from different sources (satellite, airborne and very recently by UAV). Riparian forests, one of the less represented habitats in the protected area, were assessed by 11 permanent monitoring plots where structural variables are recorded every 5 years.

2.2. Field monitoring

We carried out quinquennial inventories (summers 2004, 2010, 2014) of the composition and structure of the two foundation tree communities (*Fraxinus angustifolia* and *Salix atrocinerea*) in a permanent monitoring network established in 2004 within the La Rocina floodplain forest area (Fig. 1). We conducted tree inventories in 11 175–250 m^2 permanent plots (KML Map 1) conforming to the recommended sampling area range for wetland forests (van der Maarel, 2005). In each plot, we identified all trees higher than 1.5 m to species; we recorded species abundance in percent cover to determine species dominance; we measured tree height with a Blume-Leiss hypsometer to the nearest 0.5 m and breast height diameter (DBH) for all stems $\geq$2.5 cm to the nearest 0.1 cm at two

perpendicular directions using a caliper. For trees with single or multiple trunks at breast height, all stems were counted, measured and recorded as dead or alive. In every plot, we identified tree seedlings to species and counted them across the three sampling campaigns, recorded the presence of vegetative reproduction and made observations about tree damage due to flooding/wind and herbivory. We recorded local environmental variables such as water level in summer and descriptive observations on soil parameters (Supplemental Table S1).

During the 2014 field campaign, a specific measurement design (KML Map 2) was adopted to determine an extensive collection of homogeneous canopy covers dominated by the two target species. The complete study area was gridded at a 30 × 30 m pixel size from which a final set of 42 pixels was selected to show the highest spectral homogeneity (see details in the following section). In the field, once the center of the pixel was located and the dominant canopy cover confirmed, we recorded structural information on the forest, such as plant and bare soil percent cover, forest stratification and species dominance per strata, tree recruitment and tree damage. We took measurements at the four cardinal axis points where the canopy cover mixed with the other species. During this sampling we also measured Leaf Area Index (LAI) using AccuPAR LP-80 ceptometer (Decagon Devices, Pullman, WA, USA). LAI variability contributes to spectral response (Jones and Vaughan, 2010), sufficiently enough to produce differences for canopies homogeneously covered by the same species.

Plots and measurements were geolocated by means of a Leica 1200 differential GPS allowing an average precision positioning with an error below 1 m under forest canopy. In addition, several trees were tagged and marked to improve plot identification for later visits.

2.3. Remote sensing data sources

The long-term ecological monitoring program incorporates a landscape scale approach by using different sources of remote sensing imagery. For this study we used airborne hyperspectral images acquired with the Airborne Hyperspectral Scanner (AHS) in 2004 and 2015. These flight campaigns were originally designed for aquatic plant mapping, wetland delineation and water quality assessment of the Doñana marshes and lagoons (Gómez-Rodríguez et al., 2008; Bustamante et al., 2009; Díaz-Delgado et al., 2010, 2016). La Rocina images were also available with a 5.8 m pixel size. The AHS is a "line-scanner" radiometer and collects at-sensor radiance from the ground in 80 spectral bands from 430 to 13,000 nm. It has a wide coverage in the Visible (VIS) and Near InfraRed (NIR) regions from 430 to 1000 nm with 21 relatively wide bands (27–30 nm). In the Short Wave InfraRed (SWIR) it has a single band centered on 1600 nm. Images were acquired and preprocessed by INTA (Instituto Nacional de Técnicas Aerospaciales). They were radiometrically corrected to at-sensor spectral radiance and geometrically corrected using IMU data and the Andalusian GPS reference network with a mean position error (RMS) below 1 pixel. During each flight campaign, spectroradiometric signatures were collected in the field for bright and dark targets for use in empirical line calibration, providing accurate surface reflectance for every pixel (Bustamante et al., 2013).

2.4. Image selection and digital classification procedure

Based on *in situ* data acquisition, we selected AHS imagery acquired during spring in 2004 and 2015 to map the foundation species. Both dates cover the maximum monitored time span. Data gathered in the sampling plots (2004 and 2014) were used as training areas and ground-truth for digital image classifications. 70% of the ground-truth sample for every date was randomly selected as training pixels and the remaining 30% as test pixels for subsequent accuracy assessment.

An exploratory analysis was carried out to locate highly dominated stands for the two target species. A study area mask was built by combining the polygons classified as "Riparian Forest" from the most up-to-date land use cover map (Moreira et al., 2008) and the pixels only with NDVI (Normalized Difference Vegetation Index) values greater than 0.7 (Sobrino et al., 2012; Jiménez et al., 2005). The main reason for this latter criterion was to avoid the inclusion of the other relatively abundant surrounding tree/shrub species along the border of the protected area which were also apparent in the ortophotographs used for the land use cover map.

A final set of 20 spectral bands covering the range between 400 and 1000 nm was used for the digital image classification. The 2 airborne strips covering the La Rocina study area were merged by normalizing the overlapping pixels (Hongjian et al., 2004). Merged strips from the two different dates were segmented in order to assess the radiometric homogeneity of ground-truth plots (Chang, 2013). The final sample size was increased by including those adjacent pixels assigned to the same segments of the original training areas. The maximum likelihood method was used for the supervised per-pixel classification for both dates. Although most of the fluvial course of La Rocina is covered by a dense canopy, pure stands dominated by each of the target species are not frequent, while mixed canopies from the two target species are typical. Nevertheless, we decided to use *Salix* and *Fraxinus*-dominated classes as thematic classes, in order to reveal subtle dominances even under mixed canopies circumstances. A canopy cover gradient of this nature may subsequently be assessed with the resulting rule images and refining *a posteriori* probabilities (Strahler, 1980). Results were assessed by calculating overall agreement and Kappa index with the unused 30% of pixels as the validation test set.

2.5. Data processing and calculations

Structural data on foundation trees were organized in matrices for the calculations of structural indicators at the individual (number of stems/tree, basal area/tree) and plot level (number of stems, trees and seedlings). Using the averages of the two diameter measurements per tree, basal area per individual (cm^2/tree) was calculated for all live and dead stems. Structural indicators were calculated for each species: health category (dead/alive), dominance types (based in 2004 species dominance for each plot), and height and diameter size classes (Supplemental Table S1).

We explored existing relations in the structural plot parameters with environmental variables using correlation analyses. We used the non-parametric Kruskal Wallis Rank test to detect differences among the different structural indicators for individuals and plots by sampling years, species and dominance types (*Fraxinus* or *Salix* dominated plot), having set a significance level at $p < 0.05$.

3. Results

3.1. DPSIR framework

The main drivers, corresponding pressures and their effects on the La Rocina floodplain forests during the decade 2004–2014 are summarized in Table 1. For the most part they relate to the intensification of agriculture and reduced discharge due to eucalyptus plantations in the catchment. In parallel, some societal responses are themselves acting as drivers (management measures, gradual closure of illegal water extraction) with different impacts on the state of the ecosystem. Both external drivers and societal responses are potentially reflected through biological indicators that can be

Table 1
Summary of the main environmental and anthropic forces influencing Doñana floodplain forests during the last decades within the Driver-Pressure-State-Impact-Response (DPSIR) framework, and the corresponding indicators used for monitoring of foundation trees at both field plots (FP) and remote sensing (RS). Fx stands for *Fraxinus angustifolia,* Sa stands for *Salix atrocinerea.* [1] Iremonger and Kelly, 1988; [2] Rodríguez-González et al., 2010; [3] Stella et al., 2006; [4] Bellingham, 2001; [5] Amlin and Rood 2003; [6] Stella et al., 2010; [7] Soriguer et al., 2001.

Driving forces	Pressures	Estimated period	Environmental STATE	Impacts on ecosystem functions and foundation trees	Foundation tree indicators used in monitoring and expected change	Reference
Market demand of paper and pulp	Eucalypt plantations in the catchment.	1950–2002	Reduced discharge	Reduced hydroperiod Competitive advantage of Fx face to Sa	FP: Sa tree density drop/ Unbalanced population structure in Sa RS: Dominance shift from Sa to Fx	[1], [2]
Common Agricultural Policy subsidies for intensive farming and increased massive Tourism	Water pumping extractions increase in the surrounding area	1990–2016	Alteration of hydrologic regime (amount and timing)	Lack of seed recruitment due to loose of synchrony between species phenology (seed dispersal) and habitat availability for recruitment	FP: Seedlings monitoring/ Reproductive strategy shift from sexual to vegetative RS: Dominance shift Sa to Fx	[3], [4]
Global warming: Extreme drought climatic episodes	Rainfall scarcity	1979–1980, 2004–2005, 2012–2013	Reduced water availability	Water table drop Tree Mortality	FP: Seedlings mortality both species/Decline in both species RS: Dominance shift from Sa to Fx	[5], [6]
Societal Responses	**Pressures**	**Estimated period**	**Environmental state**	**Impacts on ecosystem functions and Foundation trees**	**Foundation tree indicators used and expected trend**	**Reference**
Conservation measures at the catchment scale	Clearing of eucalyptus plantations in the catchment	2002–2004	Increased discharge	Larger hydroperiod, increased hydrology and sediment dynamics, creation of available habitats Increased competitive advantage of Sa face to Fx	FP: Increased Sa stem density and stems/tree (recovery through vegetative sprouting) RS: Dominance shift Fx to Sa	[1], [2]
Conservation measures within the protected area	Increase of wild herbivores populations	1984–2016	Overgrazing within floodplain forest	Breakage and trampling, Recruitment elimination and browsed vegetative sprouts	FP: Lack of seedlings/ Senescent Sa populations RS: Dominance shift from Sa to Fx	[7]

tested through the monitoring program in both the field and remote sensing methods.

3.2. Field monitoring

3.2.1. Plot indicators and water level

The exploratory analysis between the plot structural parameters and the water level measured in summer in the different sampling years resulted in some significant correlations. In *Salix*-dominated plots (Sa-plots), *Fraxinus* basal area per ha was significantly correlated ($r = 0.5518$, $p = 0.0330$) with the increasing depth of the phreatic level. Meanwhile in *Fraxinus*-dominated plots (Fx-plots), no significant relationship was revealed ($r = -0.1130$, $p = 0.7266$). This confirms the ecological preferences of both species and suggests that a lower phreatic level favors the increase of *Fraxinus* in Sa-plots. Sa-plots show a significant increase of dead stems per ha ($r = 0.5524$, $p = 0.0327$) with the increasing depth of the phreatic level, while Fx-plots displayed no significant relationship ($r = -0.4735$, $p = 0.1200$). This suggests that in Sa-plots the lowering of the phreatic level may be related to increased mortality of *Salix*.

3.2.2. Individual level indicators: tree biomass and health patterns across species

We recorded a significant ($KW-H_{(2, 204)} = 16.5458$, $p = 0.0003$) decrease in the number of *Salix* stems by individual in the period 2004–2014 with a particularly large decrease in the 2004–2010 sampling (Fig. 2), while the number of *Fraxinus* stems by individual was relatively constant across the 10 year period of monitoring ($KW-H_{(2, 182)} = 0.5199$; $p = 0.7711$). The percentage of dead basal area per tree experienced a marginally significant increase ($KW-H_{(2, 204)} = 4.5094$; $p = 0.1049$) for *Salix* individuals, especially for 2014, while no significant change ($KW-H_{(2,182)=}0.2843$; $p = 0.8675$) was detected in *Fraxinus* individuals. Stems per tree can be considered as a proxy for tree biomass and the dead basal area per tree as a proxy for health status (basal area corresponding to the dead stems in a tree).

3.2.3. Plot level indicators: temporal trends of stem density, tree density and mortality

Stem and tree density (total number of stems/ha and total number of trees/ha) displayed a similar temporal pattern in Fa-plots, with a non-significant decrease in 2004 sampling compared to 2010 and subsequently was relatively constant (stems/ha: $KW-H_{(2,12)} = 1.2007$, $p = 0.5486$; trees/ha: $KW-H_{(2,12)} = 0.4865$, $p = 0.7841$) (Fig. 3). However, in Sa-plots, tree and stem density followed different trends. While the stems/ha displayed a significant decrease in the 2004–2010 sampling ($KW-H_{(2,15)} = 8.094$, $p = 0.0175$), the trees/ha showed an initial small decrease but a slight recovery in 2014 ($KW-H_{(2,15)} = 0.3834$, $p = 0.825$). This recovery of the total number trees may be related to the recruitment of *Fraxinus* individuals in Sa-plots (see Fig. 4).

3.2.4. Population structure indicators: size class frequency changes

The stem diameter distributions varied by species in the two types of plot dominance with *Fraxinus* stems present in both Fx-plots and Sa-plots, though with different frequencies, while *Salix* stems were exclusively present in Sa-plots with a negligible presence in Fx-plots (Fig. 4). In Sa-plots, the frequency of *Fraxinus* stems/ha increased especially across the 2010 and 2014 campaigns. In Sa-plots, the frequency of *Salix* stems/ha experienced a decrease from 2004 to 2010, but was unchanged from 2010 to 2014. A striking pattern in the size frequency distributions of *Salix* stems in Sa-plots is that rather than moving to bigger size classes, the total number of stems substantially decreased in the 2010 and 2014 sampling campaigns, compared to 2004 stem frequencies, probably

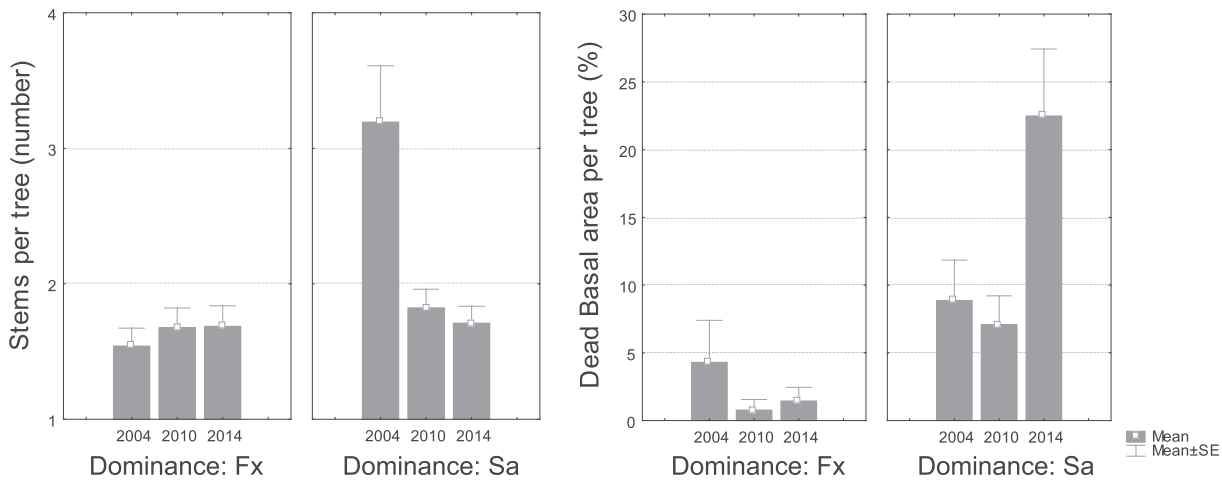

Fig. 2. Number of stems per tree (left) and percentage of dead basal area per tree (right), calculated from the data collected at the permanent plots during each of the three monitoring campaigns in 2004, 2010 and 2014. Types of dominance (Fx, *Fraxinus angustifolia*; Sa, *Salix atrocinerea*). NOTE: plot dominance based on the 2004 sampling.

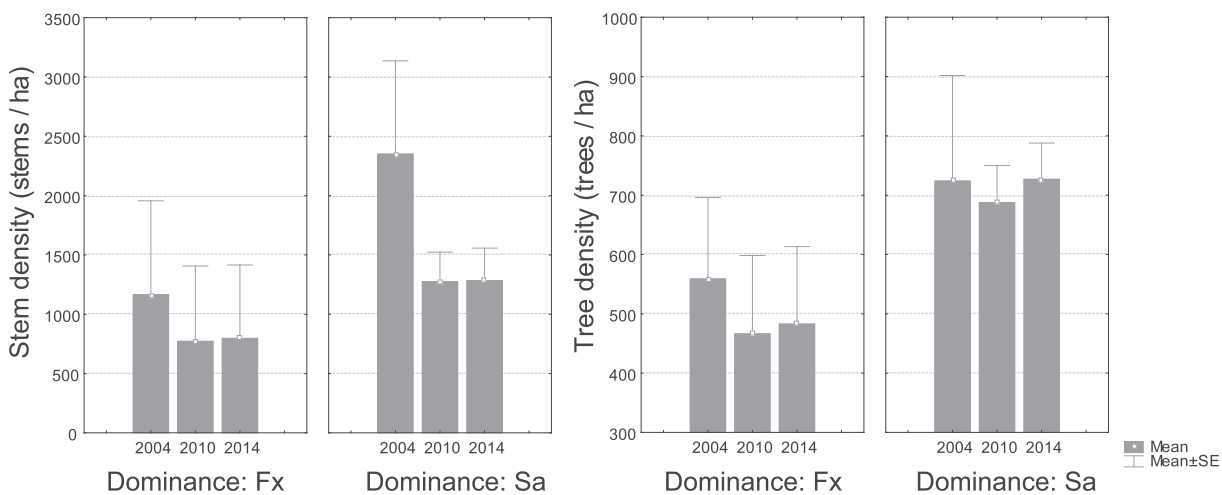

Fig. 3. Number of total stems per ha (left) and number of total trees per ha (right), calculated from the data collected at the permanent plots during each of the three monitoring campaigns in 2004, 2010 and 2014. Types of dominance (Fx, *Fraxinus angustifolia*; Sa, *Salix atrocinerea*). NOTE: plot dominance based on the 2004 sampling.

due to increasing mortality rates. Finally, there is an unusually high frequency of *Fraxinus* stems in one particular size class (27.5 cm in 2004) that is evident across the three sampling campaigns. This probably corresponds to a cohort of even-aged trees, suggesting a past recruitment event for this species.

3.2.5. Regeneration of Salix and Fraxinus in monitored plots

Salix displayed no sexual regeneration, excluding a punctual occurrence in one plot in 2004. In contrast, *Fraxinus* seedlings were present in both types of plot dominance, particularly in recent years. *Fraxinus* seedlings significantly increased ($KW{-}H_{(2,15)} = 9.5913$, $p = 0.0083$) in Sa-plots in 2010 and 2014 relative to 2004, and they were present across all Fx-plots from the beginning of the monitoring, with a non-significant increasing trend ($KW{-}H_{(2,12)} = 0.2740$, $p = 0.8720$). (Fig. 5).

3.3. Remote sensing analyses

The first exploratory analysis of spectral signatures revealed significant differences between *Fraxinus* and *Salix* dominated pixels. Differences were found across the whole infrared spectrum even for dominated canopies with a high variability in LAI (Supplemental Fig. S2). Accordingly, the AHS bands are considered as suitable for the classification of the two target species.

The resulting classifications for the two selected dates showed overall agreement values of 85.48% (2004 image) and 94.32% (2015 image). Kappa indices of agreement reached 0.70 and 0.88 for 2004 and 2015 maps, respectively. Because these accuracy values are considered as very acceptable we decided to use both maps in assessing species dominance change.

In order to highlight changes between the two dates, a change image (Fig. 6) was calculated revealing the 4 classes of change: 1) From *Salix* to *Fraxinus*; 2) From *Fraxinus* to *Salix*; 3) *Fraxinus* in both dates; 4) *Salix* in both dates.

Study of the change image (Fig. 6) reveals an overall shift from *Fraxinus*-dominated pixels to *Salix*-dominated pixels (a total of 17.66 ha from 2004 to 2015 corresponding to 14% see Table 2). The proportion of pixels dominated by *Fraxinus* declined from 56% to 48% while for *Salix* it increased from 43% to 51%. However, shifts in the opposite direction, from *Salix* to *Fraxinus,* are also ubiquitously found across the floodplain forest (a total area of 30 ha corresponding to 33% of the 2004 *Salix*-dominated area). Among the permanent plots inventoried, which were integrated in the ground-truth, two (DAM6 and DAM8) revealed the most significant trend

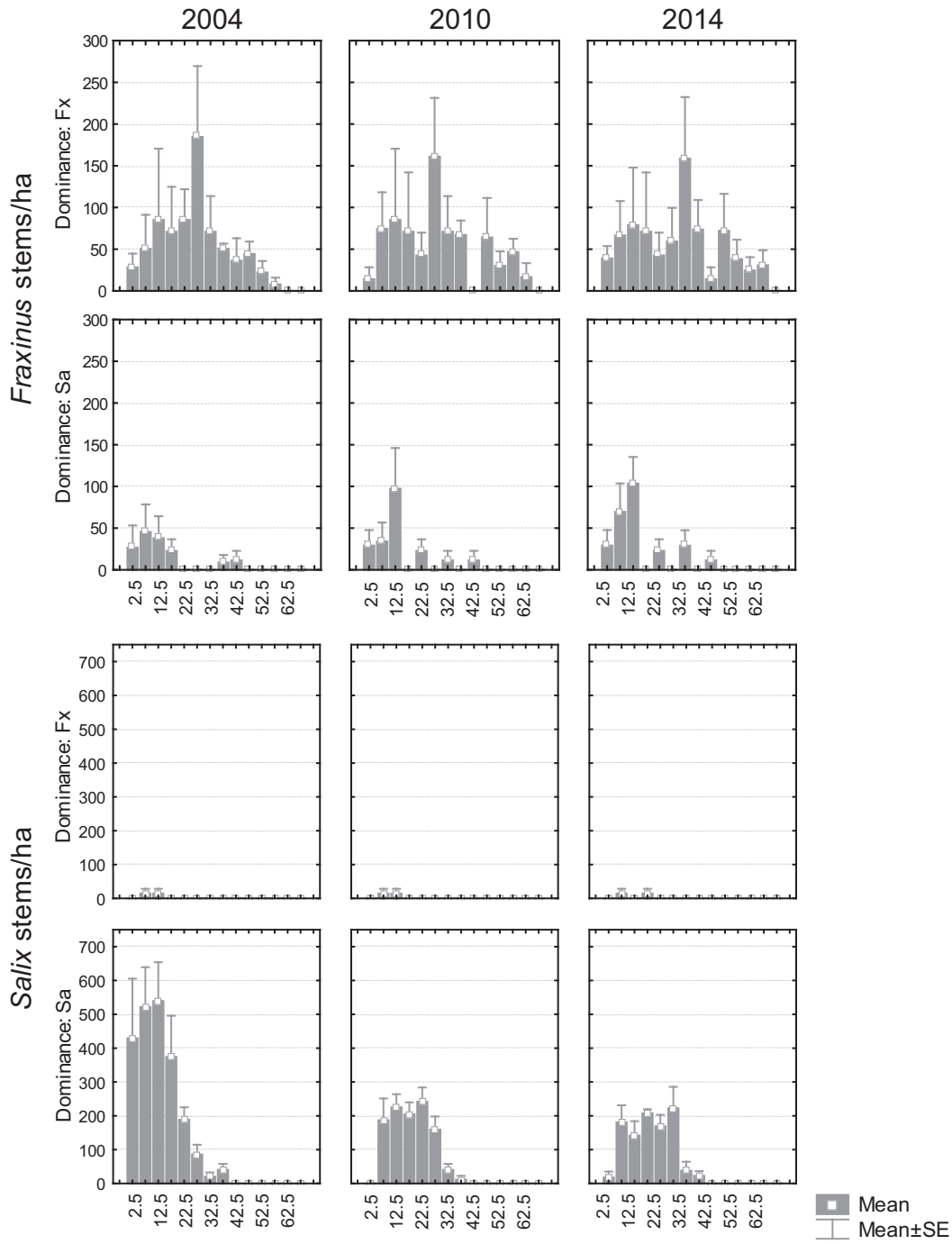

Fig. 4. Size distribution of total stems per ha, calculated from the data collected at the permanent plots during each of the three monitoring campaigns in 2004, 2010 and 2014. Types of dominance (Fx, *Fraxinus angustifolia*; Sa, *Salix atrocinerea*). NOTE: plot dominance based on the 2004 sampling.

changes, displaying a net cover change in dominance from class *Salix* to *Fraxinus* from 2004 to 2014.

Spatial arrangement of dominance shifts reveals local trends such as an increase of *Salix* dominance (blue and red colors in Fig. 6) downstream and in the uppermost zone of the study area. On the other hand, *Fraxinus* dominated pixels (orange and green colors in Fig. 6) are mainly found on the intermediate riverine course. In terms of the spatial pattern found for both species and considering their habitat preferences (that is, proximity to the water course for *Salix*, and less flooded areas for *Fraxinus*), *Salix* was identified in the expected position at the water front, in certain sections only.

4. Discussion

The present case study illustrated the impact of recent land use changes and water extractions at the La Rocina catchment on the Doñana floodplain forests, and was corroborated through forest responses at both the local (*in situ* plots) and landscape scale

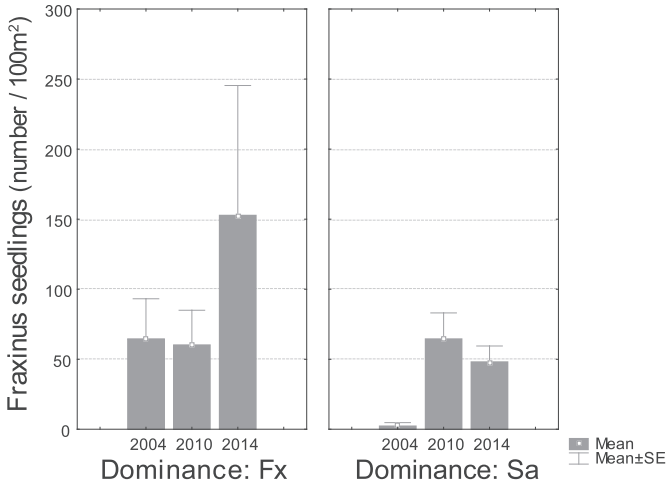

Fig. 5. Relative frequency of *Fraxinus* seedlings per 100 m^2 in *Fraxinus* dominated plots and *Salix* dominated plots from the data collected at the permanent plots during each of the three monitoring campaigns in 2004, 2010 and 2014. Types of dominance (Fx, *Fraxinus angustifolia*; Sa, *Salix atrocinerea*). NOTE: plot dominance based on the 2004 sampling.

management guidelines.

The most obvious pattern revealed by the *in situ* monitoring is a general senescing trend in the tree populations across the permanent plots. For *Salix* particularly, this pattern was revealed by the absence of young trees, a decline in the number of stems per tree, an increase in dead basal area per tree and a decreasing shift in the frequency of smaller diameter distributions that was practically reduced by half from 2004 to 2010. In parallel, a slight increase in *Fraxinus* can be observed across the studied plots, especially through recent increases in *Fraxinus* seedlings and trees with a smaller diameter in the plots that were dominated by *Salix* in 2004. These structural trends reflect the contrasting ecological and reproductive traits of *Salix* and *Fraxinus*. The absence of small *Salix atrocinerea* trees and seedlings may be the result of low seed longevity (Karrenberg et al., 2002) combined with large flooding periods during seed dispersal in early spring (Rodríguez-González et al., 2010), and competition for light with mature trees (Jones and Sharitz, 1998) due to their low shade tolerance (Battaglia and Sharitz, 2006). In addition, damage by herbivores looking for the fresher conditions within the forest in the hotter months can pose an additional challenge to the few recruitment opportunities that exist for this species (field observations). In conditions of poor

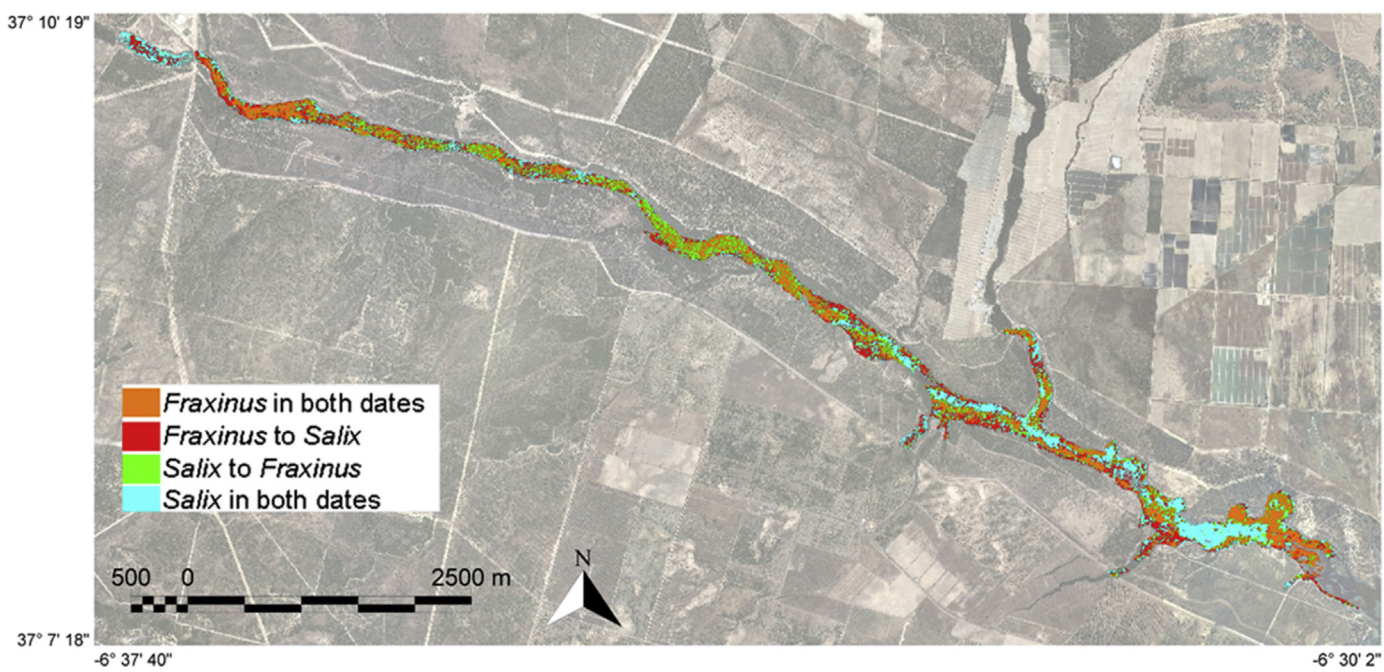

Fig. 6. Map of changes in species dominance from the comparison of 2004 and 2015 Airborne Hyperspectral Scanner image classifications. (For interpretation of color in this figure legend, the reader is referred to the web version of this article.)

Table 2
Total area covered by every target species according to digital classification of AHS images in 2004 and 2015.

	Area in 2004 (ha)	Area in 2015 (ha)
Fraxinus angustifolia	107.25	88.43
Salix atrocinerea	61.61	80.43

(remote sensing). The conceptualization of the main pressures and impacts on the target ecosystem by means of the DPSIR framework and its integration with monitoring results revealed useful to review and formulate informed responses as conservation

seedling recruitment, sprouting through long-lived multistemmed genets is assumed as a key strategy in the persistence of *Salix* populations (Bond and Midgley, 2001; Rodríguez-González et al., 2010). The different dynamics of *Salix* stem density (showing significant temporal changes across sampling years) compared with the relatively constant values of tree density observed across the studied period exemplifies this result (Fig. 3).

The correlations between *Salix* stem mortality and depth of summer water level illustrates the reduced tolerance of *Salix atrocinerea* to summer drought (Iremonger and Kelly, 1988; Rodríguez-González et al., 2010), also reported for other riparian *Salix* species (Garssen et al., 2014; Stella et al., 2010). In addition, the

correlations between the *Fraxinus* basal area and the increasing depth of the summer water level in the initially *Salix*-dominated plots suggest a forest succession in response to water retreat, i.e. a terrestrialization tendency. The natural successional dynamics characteristic of wetland forests includes drier and wetter periods (Pokorný et al., 2000), although exogenous driving factors acting at different scales might generate unexpected ecosystem trajectories (Dufour and Piégay, 2009). Consistently across several structural indicators, the largest *Salix* decline was observed from 2004 to 2010, while 2014 sampling generally showed similar values or a slight recovery from 2010. This could reflect a mixture of factors influencing water availability in the floodplain, such as a lag-effect of the past anthropic impacts (water extractions) in the catchment, coupled with growing variability of the Mediterranean climate regime. An extremely dry climatic episode during the hydrologic year 2004–2005, when total annual rainfall was only 173 mm probably imposed water limitations on floodplain species, similar to the extensive die-off reported on shrubland communities (Lloret et al., 2016). Since this intense decrease in the water level may have provoked damage in *Salix* tree populations, especially in the first years after 2005, these effects would be evident in the structural parameters of the 2010 with respect to 2004 sampling.

Results at the plot scale reveal a progressive increase in *Fraxinus* while at the landscape scale the net process is the opposite. Such discrepancies are not uncommon when scaling up *in situ* data to larger scales (Hufkens et al., 2008; Wu and Li, 2009), particularly when the type of monitored parameters are different (canopy cover *vs.* understory cover measurements). While setting up the Doñana Monitoring program, efficiency and accessibility criteria were critical to guarantee long-term monitoring (Moreira et al., 2008). The final decision on the number and location of permanent plots was subject to the trade-off between covering the 12 km long floodplain forest and the representativeness of plots dominated by each of the target species. As mentioned above, the La Rocina floodplain forest is mainly composed of mixed canopies with subtle gradients tightly related to water table depth. The sampling design focused on locating the most stands dominated by the target species. It is plausible that such structural characteristics have an effect on species mapping. However, the inclusion of mixed canopies in the training area for the digital classifications did not improve results. Nevertheless, assigning pure species classes to every single pixel has demonstrated the trends and shifts in a spatially explicit way and revealed the scale of these processes. Thus, a recent hydroperiod reconstruction for the Doñana marshes (Díaz-Delgado et al., 2016) confirms the increase at the mouth of the La Rocina tributary due to sand deposits from the El Partido stream which enters from the north (García-Novo et al., 2007). Flood duration at the mouth of La Rocina may well be the reason for the spread of *Salix* in this area (Fig. 6).

Landscape scale analysis contributes to the interpretation of the shifts in species dominance in the decade 2004–2014 by revealing different trends along the fluvial course. A detailed assessment of the trends indicates the synergetic action of the different environmental drivers determining water availability for the two target species. Manzano et al. (2005) provided the most coherent time series of piezometric values in the La Rocina stream, which presents a consistently negative trend from the 1970s (Supplemental Fig. S3). However, by the mid-1990s water discharges started to rise again. This is the starting period of catchment management where extensive mature eucalyptus plantations were removed and treated as a general restoration action (until 2011). It is likely that such eradication actions contributed to the release of water retained by eucalyptus plantations. The time span covered by our analyses assumes the indicative character of the foundation species in water availability. However, progressive reduction of water availability will favor the spread of *Fraxinus* species compared with an overall decline of *Salix* species. This has been the situation since the 1960s for La Rocina. Indeed, diameter size distributions in *Fraxinus* observed in the permanent plots suggest the occurrence of favorable conditions for *Fraxinus* recruitment within the last three/four decades. Both long-living species in such a seasonal stream will certainly subsist and endure up to certain limits. With the closure of many pumping stations and the removal of eucalyptus, water discharge has recovered. This may have contributed to the spread of *Salix* species. Given the lack of *Salix* seed recruitment, this remotely detected cover expansion is likely to be related with the vegetative sprouting behavior of *Salix* from previously existing trees (Rodríguez-González et al., 2010; and field observations across the whole floodplain area). This entails a long-term site occupancy strategy of the species, through a complex spatial dynamic among long-lived genets, which are able to expand across relatively large areas through branching ramets (Bellingham, 2001). On the other hand, the seasonality of flooding has a profound impact on sediment dragging which alters river banks and course profiles (field observations) and knocks down ramets and genets, which would re-start the sprouting process (Corenblit et al., 2011). The tremendous variability in changes found across the floodplain forest probably reflects the delayed trend of reduced water availability, the most recent instance being linked to the release of the water immobilized by eucalyptus and pumping stations.

One aim of this case-study was to confirm the consistency of monitoring approaches based on foundation tree indicators at two spatial scales. Our results revealed the complementarity of field and remote sensing monitoring for improved understanding of temporal changes in the floodplain forest from 2004 to 2014. Following the DPSIR framework, the field sampling monitoring revealed that the most substantial impact of driving forces and pressures (water consumption in the catchment and extremely dry climatic episodes) occurred during the first half of the study (2004–2010). In the second period (2010–2014) both remote sensing and field sampling results suggest that recent societal responses (eucalyptus removal, long-term monitoring and wildlife conservation measures) have produced changes in the course of floodplain forest dynamics, but indicate different processes. Impact on the water table has been dramatic (Custodio et al., 2009) with plant dominance shifts in just 10 years. Only by applying consistent and periodic monitoring protocols can the state of the floodplain forest ecosystem be assessed and early warning signals be identified. Nevertheless, the same drivers are continuing to affect the region since intensive agriculture is still being subsidized (WWF España, 2016).

In terms of floodplain forest management, this case study illustrates how monitoring information can be used to detect changes and improve future conservation plans. Remote sensing offers the advantage of covering larger areas than field approaches, since it provides an absolute quantification of cover changes at the whole forest scale, something unfeasible through field sampling alone. It also allows for less invasive monitoring across sites with difficult or restricted access in more sensitive conservation spots within protected areas. However, optical remote sensing essentially provides information about the top of the canopy for densely forested pixels. Changes at canopy level might not necessarily reflect the patterns of change at ground-level sampling such as increasing or decreasing tree density. Indeed, the multispectral analysis used in this study did not take account of the population dynamics revealed by tree inventories, specifically the increasing senesce and vegetative sprouting behavior of *Salix* entailing a complex spatial dynamic among long-lived genets, which persisted and expanded locally through ramets. Remote sensing did not take into account the negative effects of herbivory that limit the

expansion of young stems, contributing to skewed population structures and threatening the long term viability of *Salix* populations. However, these changes were reported by the field monitoring program. Monitoring programs with frequent return intervals (e.g., 5–10 years) can provide better insight on forest dynamics than assessments of change based on sporadic revisits to areas studied several decades earlier (Nguyen et al., 2015). Conversely, as spatial changes in foundation species cover occur through time, the remote sensing approach has the potential to become a very useful tool for improving and adjusting the field monitoring network to represent all the dominance cover types and their associated diversity (i.e. adaptive monitoring, Magurran et al., 2010). To enhance information on canopy structure and understory processes, other remote sensing tools such as LiDAR (Light Detection And Ranging) and SAR (Synthetic Aperture Radar) have the ability to produce a quantitative breakthrough in improving monitoring strategies (Barbosa et al., 2016).

5. Conclusions

Overall, our results suggest that the synergetic effects of intensive land use pressures in the catchment in the last 50 years, the history of management in the protected area and recent extreme climatic episodes have modulated the foundation species population trends in the La Rocina floodplain forest case study. In particular, a recent switch in the management practices that address the control of excessive water extraction at the basin scale seems to have moderated the previously observed decline in hydroperiod and, as a result, has promoted conditions for the beginning of the recovery of wetland species such as *Salix atrocinerea* (Rodríguez-González et al., 2010). The hydrologic reconnection of the La Rocina catchment appears to have promoted the resilience of floodplain forest populations as observed by White and Stromberg (2011) on the Salt River in Arizona. However, the natural floodplain dynamics including the complete range of successional stages are far from fully recovered, highlighting the need for continuing long-term monitoring of management and environmental change effects in the catchment if we are to secure the sustainability of this valuable ecosystem. Furthermore, our case study results illustrates the utility of the DPSIR framework as a conceptual tool for analyzing the whole environmental system, and the need for combining the complementary information provided by both field and remote sensing approaches in a monitoring program to quantify biological indicator trends. In conclusion, we suggest that our three-fold approach combining the DPSIR framework at the catchment scale with long-term *in situ* and remote sensing monitoring of floodplain forests can be applied to other rivers and catchments in order to inform and improve conservation plans.

Acknowledgements

The authors would like to thank the Doñana Biological Station, Doñana Biological Reserve and Doñana Monitoring Team and especially the staff who provided help during different stages of this study, such as Jose Luis Arroyo, Miguel Ángel Bravo, Begoña Arrizabalaga, David Aragonés and Pedro Jordano, who hosted the first *in situ* research project. We also want to thank the Doñana National Park Administration, especially Maria Dolores Cobo and José Maria Galán. This work was supported by the European Commission (EU FP5 Human Potential Programme, HPRI-CT-2002-00186, ECODOCA-2004 project "Floristic composition, community structure and ecological functioning of Doñana National Park wetland woods: basic tools for conservation); by the Programa de Acceso a la Infraestructura Científica y Tecnológica Singular (ICTS-25-2010), project "Doñana wetland forests as sentinels of global change"; and by the Transnational Access to Research Infrastructures within the EC 7th Framework Programme ExpeER (grant 262060, WETFORSIG project, 2014). PM Rodríguez-González was supported by the Portuguese National Foundation for Science and Technology (FCT) through SFRH/BD/6030/2001, SFRH/BPD/47140/2008, and Investigator FCT Research Grant IF/00059/2015 and by Fundación Barrie de la Maza Europe grant. Centro de Estudos Florestais is supported by FCT through grant UID/AGR/00239/2013. We are also grateful to the Instituto Nacional de Técnica Aeroespacial (INTA), Spain, for performing the airborne campaigns and the geometric correction of the AHS images. The airborne campaigns were funded by the Spanish Ministry of Science and Innovation through the research projects HYDRA (No. CGL2006-02247/BOS) and HYDRA2 (CGL2009-09801/BOS).

Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jenvman.2017.01.067.

References

Aldaya, M.M., Novo, F.G., Llamas, M.R., 2010. Incorporating the water footprint and environmental water requirements into policy: reflections from the Doñana Region (Spain). In: Martínez-Cortina, L., Garrido, A., López-Gunn, E. (Eds.), Papeles de Agua Virtual 5. Fundación Marcelino Botín, pp. 193–217.

Amlin, N.M., Rood, S.B., 2003. Drought stress and recovery of riparian cottonwoods due to water table alteration along Willow Creek, Alberta. Trees-Struct. Funct. 17, 351–358.

Barbosa, J.M., Sebastián-González, E., Asner, G.P., Knapp, D.E., Anderson, C., Martin, R.E., Dirzo, R., 2016. Hemiparasite-host plant interactions in a fragmented landscape assessed via imaging spectroscopy and LiDAR. Ecol. Appl. 26, 55–66.

Battaglia, L.L., Sharitz, R.R., 2006. Responses of floodplain forest species to spatially condensed gradients: a test of the flood-shade tolerance tradeoff hypothesis. Oecologia 147, 108–118.

Bellingham, P.J., 2001. Resprouting as a life history strategy in woody plants. Oikos 89, 409–416.

Bond, W.J., Midgley, J.J., 2001. Ecology of sprouting in woody plants: the persistence niche. Trends Ecol. Evol. 16, 45–51.

Bustamante, J., Aragonés, D., Afán, I., Díaz-Delgado, R., 2013. Métodos de corrección atmosférica en imágenes aeroportadas hiperespectrales: una comparación en las marismas de Doñana. In: Fernández-Renau, A., de Miguel Llanes, E. (Eds.), Teledetección: Sistemas Operacionales de Observación de la Tierra, XV Congreso de la Asociación Española de Teledetección. INTA, Torrejón de Ardoz, Madrid, España, pp. 136–139.

Bustamante, J., Díaz-Delgado, R., Aragonés, D., Fernández-Zamudio, R., García Murillo, P., Cirujano, S., 2009. Using hyperspectral sensors to map the spread of the invading water fern (*Azolla filiculoides*). In: Oral Presentation at the 2nd European Congress of Conservation Biology – ECCB 2009. Presented at the 2nd European Congress of Conservation Biology – ECCB 2009, Prague, Czech Republic.

Cardinale, B.J., Duffy, J.E., González, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzig, A., Daily, G.C., Loreau, M., Grace, J.B., Larigauderie, A., Srivastava, D., Naeem, S., 2012. Biodiversity loss and its impact on humanity. Nature 486, 59–67.

Chang, C.-I., 2013. Hyperspectral Data Processing: Algorithm Design and Analysis. John Wiley & Sons, Inc., Hoboken, NJ, USA.

Corenblit, D., Baas, A.C.W., Bornette, G., Darrozes, J., Delmotte, S., Francis, R.A., Gurnell, A.M., Julien, F., Naiman, R.J., Steiger, J., 2011. Feedbacks between geomorphology and biota controlling Earth surface processes and landforms: a review of foundation concepts and current understandings. Earth-Sci. Rev. 106, 307–331.

Custodio, E., Dolz, J., Manzano, M., Alcalá, F.J., 2005. Recursos de agua en la Comarca de Doñana. Fundación Doñana 21, Sevilla.

Custodio, E., Manzano, M., del Olmo, C.M., 2009. Las aguas subterráneas en Doñana: aspectos ecológicos y sociales. Agencia Andaluza del Agua, Sevilla.

Dayton, P.K., 1972. Toward an understanding of community resilience and the potential effects of enrichments to the benthos at McMurdo Sound. In: Parker, B.C. (Ed.), Colloquium on Conservation Problems in Antarctica. Allen Press, Lawrence, Kansas, USA, pp. 81–96. Antarctica.

Díaz-Delgado, R., 2010. An integrated monitoring Programme for Doñana natural space: the set-up and implementation. In: Hurford, C., Schneider, M., Cowx, I. (Eds.), Conservation Monitoring in Freshwater Habitats: a Practical Guide and Case Studies. Springer, Dordrecht, pp. 325–337.

Díaz-Delgado, R., Aragonés, D., Ameztoy, I., Bustamante, J., 2010. Monitoring marsh dynamics through remote sensing. In: Hurford, C., Scheneider, M., Cowx, I. (Eds.), Conservation Monitoring in Freshwater Habitats: a Practical Guide and

Case Studies. Springer, Dordrecht, pp. 375–386.
Díaz-Delgado, R., Aragonés, D., Afán, I., Bustamante, J., 2016. Long-term monitoring of the flooding regime and hydroperiod of Doñana marshes with landsat time series (1974–2014). Remote Sens. 8, 775.
Dufour, S., Piégay, H., 2009. From the myth of a lost paradise to targeted river restoration: forget natural references and focus on human benefits. River Res. Appl. 25, 568–581.
EEA, 2005. The European Environment — State and Outlook 2005. European Environment Agency, Copenhagen.
Ellison, A.M., Bank, M.S., Clinton, B.D., Colburn, E.A., Elliot, K., Ford, C.R., Foster, D.R., Kloeppel, B.D., Knoepp, J.D., Lovett, G.M., Mohan, J., Orwig, D.A., Rodenhouse, N.L., Sobczak, W.V., Stinson, K.A., Stone, J.K., Swan, C.M., Thompson, J., Von Holle, B., Webster, J.R., 2005. Loss of foundation species: consequences for the structure and dynamics of forested ecosystems. Fron. Ecol. Environ. 3, 479–486.
García Novo, F., Alonso Vizcaíno, E.M., Marín Cabrera, C., 2006. Doñana-Water and Biosphere. Mateu Cromo Artes Graficas, Madrid.
García-Novo, F., García, J.C.E., Carotenuto, L., Sevilla, D.G., Faso, R.P.F.L., 2007. The restoration of El Partido stream watershed (Doñana Natural Park): a multiscale, interdisciplinary approach. Ecol. Eng. 30, 122–130.
Garssen, A.G., Verhoeven, J.T.A., Soons, M.B., 2014. Effects of climate-induced increases in summer drought on riparian plant species: a meta-analysis. Freshw. Biol. 59, 1052–1063.
Gómez-Rodríguez, C., Bustamante, J., Koponen, S., Díaz-Paniagua, C., 2008. High-resolution remote-sensing data in amphibian studies: identification of breeding sites and contribution to habitat models. Herpetol. J. 18, 103–113.
González del Tánago, M., García de Jalón, D., Román, M., 2012. River restoration in Spain: theoretical and practical approach in the context of the European water framework directive. Environ. Manage 50, 123–139.
Gumiero, B., Rinaldi, M., Belletti, B., Lenzi, D., Puppi, G., 2015. Riparian vegetation as indicator of channel adjustments and environmental conditions: the case of the Panaro River (Northern Italy). Aquat. Sci. 77, 563–582.
Haberl, H., Gaube, V., Díaz-Delgado, R., Krauze, K., Neuner, A., Peterseil, J., Plutzar, C., Singh, S.J., Vadineanu, A., 2009. Towards an integrated model of socioeconomic biodiversity drivers, pressures and impacts. A feasibility study based on three European long-term socio-ecological research platforms. Ecol. Econ. 68, 1797–1812.
Hongjian, Y., Yun, S., Shukai, L., 2004. Fast rectifying airborne infrared scanning image based on GPS and INS. Future Gener. Comp. Sy., Geocomputation 20, 1209–1214.
Hufkens, K., Bogaert, J., Dong, Q.H., Lu, L., Huang, C.L., Ma, M.G., Che, T., Li, X., Veroustraete, F., Ceulemans, R., 2008. Impacts and uncertainties of upscaling of remote-sensing data validation for a semi-arid woodland. J. Arid. Environ. 72, 1490–1505.
Iremonger, S., Kelly, D.L., 1988. The responses of four Irish wetland tree species to raised soil water levels. New Phytol. 109, 491–497.
Jiménez, M., Díaz-Delgado, R., Soriguer, R., Fernández-Renau, A., Prado, E., Gutiérrez, O., 2005. Aproximación de las imágenes hiperespectrales AHS a la cartografía de abundancia de matorral. Rev. Teledetección 24, 95–100.
Jones, R.H., Sharitz, R., 1998. Survival and growth of woody plant seedlings in the understorey of floodplain forests in South Carolina. J. Ecol. 86, 574–587.
Jones, H.G., Vaughan, R.A., 2010. Remote Sensing of Vegetation: Principles, Techniques, and Applications. Oxford University Press, Oxford, New York.
Karrenberg, S., Edwards, P.J., Kollmann, J., 2002. The life history of Salicaceae living in the active zone of floodplains. Freshw. Biol. 47, 733–748.
Lloret, F., Riva, E.G., Pérez-Ramos, I.M., Marañón, T., Saura-Mas, S., Díaz-Delgado, R., Villar, R., 2016. Climatic events inducing die-off in Mediterranean shrublands: are species' responses related to their functional traits? Oecologia 180, 961–973.
Magurran, A.E., Baillie, S.R., Buckland, S.T., Dick, J.M., Elston, D.A., Scott, E.M., Smith, R.I., Somerfield, P.J., Watt, A.D., 2010. Long-term datasets in biodiversity research and monitoring: assessing change in ecological communities through time. Trends Ecol. Evol. 25, 574–582.
Manzano, M., Custodio, E., Mediavilla, C., Montes, C., 2005. Effects of localised intensive aquifer exploitation on the Doñana wetlands (SW Spain). In: Sahuquillo, A., Capilla, J., Martínez-Cortina, L., Sánchez Vila, X. (Eds.), Groundwater Intensive Use, International Association of Hydrogeologists, Selected Papers on Hydrogeology. Taylor & Francis, pp. 209–219.
Millennium Ecosystem Assessment (MEA), 2005. Ecosystems and Human Well-being: Synthesis. Island Press, Washington, DC.
Moreira, J.M., Quijada, J., Ortega, E., Romero, D., Gil, Y., 2008. El proyecto SIOSE en Andalucía. In: Hernández-Calvento, L., Parreño, J.M. (Eds.), Tecnologías de la Información Geográfica para el Desarrollo Territorial (Jornadas Técnicas). Servicio de Publicaciones y Difusión Científica de la ULPGC. Las Palmas de Gran Canaria, pp. 941–953.
Morris, E.P., Flecha, S., Figuerola, J., Costas, E., Navarro, G., Ruiz, J., Rodríguez, P., Huertas, E., 2013. Contribution of Doñana wetlands to carbon sequestration. PLoS One 8, e71456.
Nguyen, U., Glenn, E.P., Nagler, P.L., Scott, R.L., 2015. Long-term decrease in satellite vegetation indices in response to environmental variables in an iconic desert riparian ecosystem: the Upper San Pedro, Arizona, United States. Ecohydrology 8, 610–625.
Pokorný, P., Klimešová, J., Klimeš, L., 2000. Late Holocene history and vegetation dynamics of a floodplain alder carr: a case study from eastern Bohemia, Czech Republic. Folia Geobot. 35, 43–58.
Rodríguez-González, P.M., Ferreira, M.T., Albuquerque, A., Espirito Santo, D., Ramil Rego, P., 2008. Spatial variation of wetland woods in the latitudinal transition to arid regions: a multiscale approach. J. Biogeogr. 35, 1498–1511.
Rodríguez-González, P.M., Stella, J.C., Campelo, F., Ferreira, M.T., Albuquerque, A., 2010. Subsidy or stress? Tree structure and growth in wetland forests along a hydrological gradient in Southern Europe. For. Ecol. Manage 259, 2015–2025.
Sanders, S., Kirschbaum, J., 2015. Forest health monitoring on the st. Croix national scenic riverway: short-term change and longer-term projections. J. Torrey Botanical Soc. 142, 271–282.
Sobrino, J.A., Franch, B., Mattar, C., Jiménez-Muñoz, J.C., Corbari, C., 2012. A method to estimate soil moisture from Airborne Hyperspectral Scanner (AHS) and ASTER data: application to SEN2FLEX and SEN3EXP campaigns. Remote Sens. Environ. 117, 415–428.
Soriguer, R.C., Delibes, J.R., Fandos, P., Zamora, R.Y., Rodríguez, A., 2001. Análisis de la incidencia de los grandes herbívoros en la Marisma y en la Vera del Parque Nacional de Doñana. Organismo Autónomo de Parques Nacionales. Ministerio de Medio Ambiente, Madrid.
Stella, J.C., Battles, J.J., Orr, B.K., McBride, J.R., 2006. Synchrony of seed dispersal, hydrology and local climate in a semi-arid river reach in California. Ecosystems 9, 1200–1214.
Stella, J.C., Battles, J.J., McBride, J.R., Orr, B.K., 2010. Riparian seedling mortality from simulated water table recession, and the design of sustainable flow regimes on regulated rivers. Rest. Ecol. 18, 284–294.
Stella, J.C., Rodríguez-González, P.M., Dufour, S., Bendix, J., 2013. Riparian vegetation research in Mediterranean-climate regions: common patterns, ecological processes, and considerations for management. Hydrobiologia 719, 291–315.
Strahler, A.H., 1980. The use of prior probabilities in maximum likelihood classification of remotely sensed data. Remote Sens. Environ. 10, 135–163.
Ström, L., Jansson, R., Nilsson, C., 2012. Projected changes in plant species richness and extent of riparian vegetation belts as a result of climate-driven hydrological change along the Vindel River in Sweden. Freshw. Biol. 57, 49–60.
van der Maarel, E., 2005. Vegetation Ecology. Blackwell, Cornwall.
Vörösmarty, C.J., McIntyre, P.B., Gessner, M.O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S.E., Sullivan, C.A., Liermann, C.R., Davies, P.M., 2010. Global threats to human water security and river biodiversity. Nature 467, 555–561.
White, J.M., Stromberg, J.C., 2011. Resilience, restoration and riparian ecosystems, case study of a dryland, urban river. Rest. Ecol. 19, 101–111.
Wu, H., Li, Z.-L., 2009. Scale issues in remote sensing: a review on analysis, processing and modeling. Sensors 9, 1768–1793.
WWF España, 2016. El estado del agua en Doñana. Una evaluación del estado de las aguas y los ecosistemas del espacio protegido. WWF-Informe 2016, WWF/Adena, Madrid. Available online:. Accessed 25/11/2016. http://awsassets.wwf.es/downloads/estado_del_agua_donana.pdf.