THE USE OF RIPARIAN FORESTS AS ECOLOGICAL CORRIDORS BY PASSERINE BIRDS IN THE SOUTH OF PORTUGAL

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The use of riparian forests as ecological corridors by passerine birds in the South of Portugal

Abstract

Despite riparian forests have long been considered ecological corridors still few empirical evidence exists. This thesis analyses the use of riparian forests as ecological corridors by birds in a Mediterranean type climate, focusing primarily on passerines. The aim was to investigate: 1) if riparian habitats supported significantly richer and more abundant assemblages regardless of the riparian quality and matrix; 2) if birds selected these habitats differently; 3) if habitat use changed after pressures upon riparian forests; 4) how these changes affected bird assemblages and condition; 5) which characteristics of the riparian area can be used to enhance their use as corridors and 6) which restoration practices could be developed to achieve broader conservation strategies. Point count and mist netting protocols were used in two distinct river basin of the South of Portugal (Tagus and Odelouca).

Riparian forests supported more diverse and abundant communities independently of matrix and riparian habitat quality and common European woodland bird species were using them as surrogate/refuge in depleted situations. After riparian clear-cut: richness and abundance significantly decreased in all habitats; the number of days migrants remained in the area was significantly lower and fat scores and mean weight were significant lower. To updated knowledge these findings have never been reported and suggest that poor quality riparian habitats might act as ecological traps and this should be considered when designing mitigation measures, restoring riparian corridors and developing ecological networks. Our results support the idea that riparian forests in xeric context act as corridors for resident and migratory birds.

A utilização das galerias ribeirinhas como corredores ecológicos pelas aves Passeriformes no Sul de Portugal

Resumo

Apesar das galerias ribeirinhas serem consideradas corredores desde há muito, poucos estudos empiricos evidenciam esta característica. Esta tese analisa a utlização das galerias ribeirinhas como corredores ecológicos pelas aves passeriformes num clima Mediterrânico. Pretendeu-se investigar: 1) se as galerias ribeirinhas suportam taxocenoses mais ricas e abundantes independentemente da matriz e da qualidade da galeria; 2) se as aves selecionam estes habitats de modo diferente; 3) se pressões sobre a galeria condicionam o uso do habitat; 4) como as alterações afectam a composição das comunidades e a sua condição; 5) que características da galeria podem ser melhoradas para facilitar o seu uso como corredores; 6) que práticas de restauro podem ser desenvolvidas para atingir objectivos de conservação mais vastos. Foram usados protocolos de contagem por pontos e anilhagem em duas bacias hidrográficas distintas no Sul de Portugal (Tejo e Odelouca).

As galerias observadas suportaram comunidades mais diversas e abundantes independentemente da matriz e da qualidade do habitat ribeirinho. Verificou-se que as espécies de bosque europeias usaram as galerias como refúgio em situações de depleção. Após o corte da mata ribeirinha a riqueza e a abundância decresceram significativamente em todos os habitats, o número de dias que os migradores ficavam na área decresceu significativamente e a gordura e o peso médio de algumas espécies decresceu significativamente. Estes resultados nunca foram antes registados e sugerem que habitats ribeirinhos de fraca qualidade podem ser armadilhas ecológicas, o que deve ser considerado quando se desenham medidas de mitigação, restauro e se desenvolvem redes de corredores ecológicos. Os resultados suportam a ideia de que as galerias em contexto xérico funcionam como corredores para aves passeriformes residentes e migradoras.

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Chapter 1

GENERAL INTRODUCTION

1. General Introduction

1.1 Habitat loss and fragmentation

Habitat loss and fragmentation as a result from human activities together with climate change are responsible for high extinction rates of eukaryote species at a global scale due to reduction of suitable habitat for surviving and to reach new areas for colonizing (Didham, 2010). Habitat fragmentation is the process by which habitat loss results in the division of large, continuous habitats into a greater number of smaller patches of lower total area, isolated by a matrix of dissimilar habitats (Ewers & Didham, 2006). Thus, habitat fragmentation only occurs when habitat loss reaches the point where continuity is broken (Opdam & Wiens, 2002). The proportions of patches that are altered by external conditions are termed edge habitat, while unaffected portions are called core habitat (Ewers & Didham, 2006).

Anthropogenic habitat modification is the most important large scale cause of habitat fragmentation (Fahrig, 2003) and it is considered the most important cause of biodiversity loss (*e.g.* Dirzo & Raven, 2003; Didham, 2010; Haddad *et al*., 2015). However, evidence of effects of habitat loss on biodiversity is much stronger than evidence of habitat fragmentation (Fahrig, 2003; Bennet, 2003). Habitat loss (frequently conceptualized and measured by researchers as habitat fragmentation) has effects on biodiversity, affecting species abundance, distribution, genetic diversity, growth rate, change in trophic chain, species interactions, breeding success, dispersal success, predation rate and animal behaviour (Fahrig, 2003). Habitat fragmentation is more difficult to study because it is difficult to distinguish between habitat loss and the “breaking apart” of habitat as a result of habitat loss. Farhig (2003) review on the effects of true habitat fragmentation (studies that maintained the amount of habitat constant but increased their fragmentation) found evidence for negative effects on: species richness, abundance, density, distribution, reproduction, movement and behaviour.

1.2 Ecological corridors

Based on Island biogeography theory (Mac Arthur & Wilson, 1967), Wilson & Willis (1975) have proposed the development of ecological corridor networks to mitigate the impacts of habitat fragmentation. Since then considerable research has been devoted to ecological corridors. Despite several scientists argued that experimental evidence of effectiveness of ecological corridors was virtually inexistent (Simberloff & Cox, 1987; Simberloff *et al*., 1992; Meffe & Carrol, 1994, Mech & Hallet, 2001) and even ambiguous (Nicholls & Margules, 1991; Saunders & Hobbs, 1991), empirical evidence of negative impacts is rare (Haddad *et al*., 2011). The number and accurateness of studies on the positive effects of corridors is increasing and providing evidence that corridors are valuable conservation tools (review in Beier & Noss, 1998; Tewksbury *et al.*, 2002, Haddad *et al.,* 2011; LaPoint *et al.,* 2013).

The advantages of the development of networks of ecological corridors are associated with the increase of connectivity among organisms: increase in immigration between reserves, increase in richness and abundance, decrease of inbreeding rate, increase in foraging areas, increase in habitat variety and increase in refuge availability in case of catastrophe (Noss, 1987). Moreover, corridors are likely to become more important as many species undergo range shifts in response to climate change but there is little knowledge on the use of corridors in a changing climate (Heller & Zavaleta, 2009; Sekercioglu, 2009, Krosby *et al*., 2010). On the other hand, negative effects of corridors have been associated with dispersal of antagonistic species, including predators, diseases and invasive species (Simberloff & Cox, 1987), increased edge effects and increase fixation of deleterious mutations (Haddad, 2011).

In research studies devoted to ecological corridors, a number of corridor definitions has been used depending on the background of the researcher (ecology, population biology or landscape ecology). The definition by Forman & Godron (1986) defined corridors as "*narrow strips of land which differ from the matrix on either side*". This definition is particularly important because it is rooted in the habitat fragmentation definition (the process by which habitat loss results in patches of lower total area, isolated by a matrix of dissimilar habitats), i.e. the original cause of corridor idea. In fact, this definition is somewhat retaining the "connection between habitat" not implying any function of the corridor, and it is linked to the knowledge that many species are not adapted to the surrounding habitat (matrix) (Rosenberg *et al.*, 1995). However, corridors have also been defined as continuous, narrow patches of vegetation that facilitate movement among habitat patches (Merriam, 1984).

Whereas the former definition encompasses a structural definition (form: shape-composition-context) the other introduces a function. The first definition associates habitat quality to population connectivity. The second definition has its origins in the Island Biogeography and Metapopulation theories, which include the use of corridors to enhance movement of metapopulations (Hess & Fisher, 2001) and evaluate the effects of corridors on the key population parameters (birth, immigration, death and emigration) (Haddad *et al.*, 2011).

In Hess & Fisher's (2001) review of the corridors definitions, they have gathered six ecological functions provided by corridors: habitat, conduit, filter, barrier, source and sink (Figure 1).

The following definitions were proposed:

- Conduit: organisms pass from one place to another but not reside within the corridor;
- Habitat: organisms or material can survive and reproduce in the corridor;
- Filter: some organisms or material can pass through the corridor others cannot;
- Barrier: organisms or material cannot cross the corridor
- Source: organisms or material emanate from the corridor;
- Sink: organisms or material enter the corridor and are destroyed.

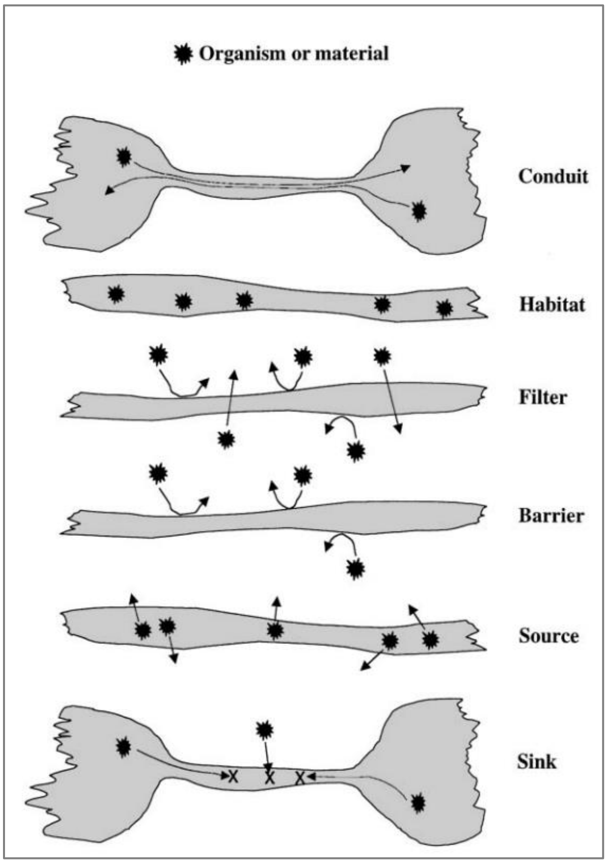

Figure 1.1 Corridor functions (adapted from Hess & Fisher, 2001).

Bennet (2003) uses similar definitions; however, the sink definition encompasses decrease in rate of mortality in the corridor from predation and other causes, which creates a net loss in the population of either corridor residents or migrant species.

In this thesis the functions of habitat, conduit, source and sink or riparian corridors are addressed, using birds as the key-organisms having in mind that the background objective of creation of ecological corridors is to improve connectivity of fragmented landscapes, contributing to overcome biodiversity loss.

1.3 Difficulties in the study of corridors

One of the main weaknesses to corridor study and planning is the missing integration between patterns of landscape composition and configuration and processes of habitat selection and movement (Cheryl-Lesley *et al.*, 2006). The study of behavioural processes of habitat selection and movement determine how animals use landscapes and thereby are fundamental to the identification and evaluation of corridors (Cheryl-Lesley *op. cit.*).

Another major problem in studying corridors is to separate connectivity from edge effects, since edge is an avoidable consequence of the creation of a corridor (Haddad *et al.*, 2011). This is problematic because responses to corridors and edges can be confounded (*e.g*. corridors or edge could increase population density) and because corridors and edges may have opposing effects on population in a source-sink context (*e.g*. corridors may increase dispersal but edges created by corridors may increase predation during dispersal) (*e.g.* Haddad *et al.*, 2011).

Forested edges are typically hotter, drier and windier than the interior of patches, with a higher light intensity and modified plant composition and habitat structure. These features affect patterns of habitat use and the relative abundance of animals at patch edges as well (Didham, 2010). Species richness typically increases at the edge and substantial turnover in species composition can occur (Ewers & Didham, 2008). High species richness at edges is a result of species influx from adjacent habitats (Didham, 2010). The quantification of edge impact requires explicit discrimination of two quite distinct components of edge influence: edge extent (i.e. the distance over which a statistical difference in response can be detected between the matrix and the patch) and edge magnitude (i.e. the degree of difference in response between the patch interior and the matrix interior) (Ewers & Didham, 2006). Also very important is the development of “two-sided” nature edge studies, because most drivers of edge effects are originated external to the patch (Didham, 2010).

1.4 Riparian habitats as ecological corridors

Biological corridors may include linear patches, such as streamside riparian areas, shelter belts, forest remnants remaining from tree harvest and, in agricultural areas, fencerows (Rosenberg *et al.*, 1997). Riparian zones in particular can serve as strips of corridors and protect important habitat at the same time, especially in more arid zones (since many species may use them complementarily). However they will not serve as corridors for species that avoid mesic conditions (Meffe & Carrol, 1994). Riparian areas are ideally suited to form the basis of linked wildlife habitat networks because they form a hierarchy of natural corridors throughout the landscape (Knopf & Samson, 1994; Bennett, 2003; Perry *et al.*, 2011); are used by most forest-dependent species (*e.g.* Jobin *et al*., 2004); and they also act as buffers to protect water quality and aquatic ecosystems (Naiman & Decamps, 1997; Cushman *et al.*, 2013). Riparian corridor connections should help to sustain wildlife populations in remnant forest patches by allowing movement between patches, while also increasing wildlife diversity within the riparian area since without connection to larger remnants, the riparian corridors themselves are also small, narrow habitat fragments (Catteral *et al.*, 2007) and contribute to landscape level diversity (Bennett *et al*., 2014).

Despite being narrow and even if likely degraded by edge effects, riparian corridors also play a role in biodiversity conservation in intense agricultural landscapes because they represent remnants of both wetlands and woody habitat once available for wildlife (Jobin *et al.*, 2004; Catteral *et al.*, 2007). Destruction of riparian vegetation causes local extinction and also reduces the ability of some populations to recolonize sites (Knopf & Samson, 1994).

Although the common assumption and generalized used of reconstructing riparian corridors as a management measure, still few quantitative data address this assumption (Naiman & Decamps, 1997). One of the major problems associated with riparian habitats and their study as ecological corridors is the occurrence of cumulative ecotonal and edge effects since riparian areas are transition zones of plant communities changing from aquatic to terrestrial systems in a relatively small spatial scale for most rivers and because they are the border of different habitats.

In this thesis the use of riparian habitats as corridors was studied, with special focus on structural and functional connectivity. The study addressed two behavioural processes - habitat selection and movements - that determine how animals use landscapes and are fundamental to the identification of corridors (Chetkiewicz *et al.*, 2006). Considering that colonisations and re-colonisations from surrounding landscapes are important to local community dynamics (Boulinier *et al.*, 2001), riparian forest spillover effect (source) was also analysed since this function maybe important to re-establish connectivity at landscape level (Didham, 2010; Damschen *et al.,* 2006; Brudvig *et al*., 2009). Finally, riparian habitat features that may lead them to act as ecological traps (sinks) (Schlaepfer *et al.*, 2002) was also studied. Bennet (2003) definition of sink was considered (increase in rate of mortality in the corridor from predation and other causes, which creates a net loss in the population of either corridor residents or migrant species).

1.5 Bird communities in riparian forests

Even though the riparian habitats have been recognized for high levels of biodiversity, 80% of the riparian corridor area of the North America and Europe has disappeared in the last 200 years (Naiman & Decamps, 1997; Popotnik & Giuliano, 2000). Nearly 70% of vertebrate species in a region will use riparian corridors in some significant way during their life cycle (Raedeke, 1989 in Naiman *et al*, 1993). Near-river habitats support a characteristic species assemblage which has practical consequences for river and riparian conservation and restoration (Sabo *et al,* 2005).

Due to their conspicuousness, birds are one of the best studied groups that occur in riparian areas. Most studies have shown higher bird species richness and abundance in riparian zones (Saunders & Edge, 1998; Whitaker & Montevechi, 1999; Jobin *et al.*, 2004; Palmer & Bennett, 2006; Cooke & Zack, 2009), however, it seems that this characteristic does not hold in mesic landscapes, where forests structural features are generally continuous between riparian and upland habitat (Murray & Stauffer, 1995; Whitaker & Montevecchi, 1997).

Several factors affect bird diversity and abundance in riparian forests: dominant tree species (Strong & Bock, 1990, Powell & Steidl, 2000), riparian width (Hagar, 1999; Pearson & Manuwal, 2001; Scott *et al*., 2003; Cooke & Zack, 2009), height (Cooke & Zack, 2009), area (Groom & Grubb Jr, 2002), adjacent upland vegetation (Strong & Bock, 1990), stream geomorphology (Scott *et al*., 2003), insect prey (Iwata *et al.*, 2003) and hydrological characteristics (Vaughan et al., 2007). While some researchers found that plant species composition (diversity and heterogeneity) affect bird diversity and abundance (Sanders & Edge, 1998; Powell & Steidl, 2000; Scott *et al.*,

2003) others did not found any relationship between riparian bird communities and plant species richness or dominance of non-native plants, provided that vegetation community retains sufficient structural diversity (*e.g.* Fleishman *et al.*, 2003).

The higher bird biodiversity richness and abundance that riparian areas exhibit has been attributed due to the juxtaposition of two habitat types (aquatic and terrestrial), which causes edge effects (Meffe & Carroll, 1994; Whitaker & Montevecchi, 1999). Among the reported effects of edge on bird communities are: high rates of nest predation (Paton, 1994; Powell & Steidl, 2000), population decline in some species (Yahner, 1988), or increase (Whitaker & Montevechi, 1999). Edge is also influenced by the matrix (Strong & Cook, 1990; Kilgo *et al.*, 1998). However, matrix effects in riparian areas seem to show some dependency on species characteristics (Perry *et al.*, 2011).

Although some researchers mention that corridors can work better for all taxa, except for birds due to their ability to fly over larger section of unsuitable habitat (Gilbert-Norton *et al.*, 2010; Cushman *et al.*,2013), recent studies have demonstrated the importance of riparian areas as ecological corridors for specific bird species in the tropics (Gillies & St. Clair, 2008; Sekercioglu, 2009) and for communities of American resident and migratory birds (Skagen *et al.*, 1998; Means & Finch, 1999; Finch & Young, 2000; Skagen *et al.*, 2005). Hass (1995) found that birds dispersed preferentially using riparian continuous corridors. Matchans *et al.* (1996) demonstrated that riparian buffer strips enhanced movements of juveniles during dispersal (i.e. acted as corridors) and maintained movements rates of adults, while referring the existence of a possible threshold distance between reserves below which birds may be less reluctant to fly across openings, making corridor use less important. Bentley & Catterall (1997) have explored the role of riparian areas as habitat corridors for birds and registered that riparian bushland supported higher richness and abundance than dryland. Skagen *et al*. (1998) concluded that all riparian patches in southeastern Arizona are important as stopover sites to en route migrants regardless of their site of isolation or connectivity. Despite these findings still few quantitative studies clearly demonstrate the role of rivers as ecological corridors for birds namely in the European context. In this thesis we focused on the role of rivers as ecological corridors for birds particularly their habitat, conduit, source and sink functions that improve connectivity in fragmented landscapes and increase biodiversity.

1.6 Riparian forests in the Mediterranean Region

In Southern Europe and the Mediterranean Region in particular, high levels of human development occur and climate changes are increasing annual average air temperatures, decreasing annual average precipitation, causing hydrologic alterations and increasing in frequency, intensity and duration of extreme events (Filipe *et al*., 2013). The development of network of ecological corridors is even more important since isolated populations are at risk of local extinctions (Cuttelo *et al.*, 2008) for example as a result of fire and drought. Movement and re-colonization can be aided by networks of riparian corridors across landscapes since they are a distinct habitat from the matrix, often not burnt during fires (Pettit & Naiman, 2007) and support higher diversity than the matrix (Stella *et al.*, 2013).

Several studies have documented the high bird species diversity and abundance in the Mediterranean riparian ecosystems (Décamps & Décamps, 2002; Blondel, 2003; Rabaça, 2004; Godinho *et al*., 2010; Pereira *et al*., 2014). Birds react to the vegetation complexity and structure in Mediterranean riparian areas (Rabaça, 2004; Godinho *et al.*, 2010) and the matrix seems to have influence on riparian bird communities (Pereira *et al.*, 2014). However, little is known on the importance of water courses as ecological corridors by passerine birds in the Mediterranean region. This hypothesis was studied by Catry *et al.*, (2009) but no significant migratory corridor along the lower Guadiana during summer-autumn migration was detected. However, observation points were used to measure riparian habitat use by migratory passerine birds, which is not the most appropriate method to detect passerine birds in riparian ecosystems, in a life cycle period where vocalizations are scarse (Karr, 1981).

1.7 Objectives

This thesis aims to evaluate the use of riparian areas as ecological corridors by songbirds in the Mediterranean region namely in Southwestern Iberia. Several corridor functions were analysed and effects on the ability of the riparian forest to promote connectivity and decrease habitat fragmentation were explored. The main focus was on songbirds, during breeding and migration periods, to explore different corridor functions of riparian habitats. This thesis had the following specific objectives:

1. Evaluate if we can characterize bird assemblages of riparian galleries in European Mediterranean context.
2. Analyse how the different bird assemblages react to degradation of the riparian area;
3. Evaluate the effect of environmental changes of riparian corridors in bird assemblages;
4. Identify the characteristics of the riparian area that can be used to enhance their use as ecological corridors for birds;
5. Contribute to the development of restoration measures to apply in Mediterranean riparian rivers that enhance bird biodiversity in fragmented landscapes.

Chapter 2 explores if we can distinguish a set of bird species that is using rivers as habitat corridors. The study intended to investigate if riparian habitats supported significantly richer and more abundant assemblages regardless of riparian quality and matrix surrounding. Our aim was also to determine whether we could demonstrate that riparian forests in xeric environments were being used as habitat refuges and corridors, therefore it addresses objectives 1 and 2.

Chapter 3 evaluates the role of several organisms (including birds) as a reliable links between organisms and environmental factors and physical disturbance (pressures) of the wider river corridor (habitat function), analyzing how these different communities react to changes in the riparian corridor and how this influences the bird community composition (addressing objective 3 and 4). This chapter was published in *Freshwater Biology 54*: 2383–2400 (2010).

Chapter 4 focus on conduit riparian corridor function and how bird assemblages react when riparian forests become absent (objective 3). This chapter was recently submitted to *Biological Conservation*.

Chapter 5 addresses if birds, among other biological assemblages, can be used as reliable indicators of restoration measures in Mediterranean rivers (objective 5). This chapter was published in *Environmental Management 46 (2)*:285-301 (2010).

Finally, chapter 6 discusses the role of riparian forests as ecological corridors for birds.

The author of the thesis clarifies that some works developed are already published and they were carried out in co-authorship with other researchers. In all of the works presented the author participated actively in the conceptualization, collection and analysis of data. Some of the guidelines of the magazines where the manuscripts were published were retained in this thesis.

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Chapter 2

LIVING ON THE EDGE: THE EFFECT OF HABITAT QUALITY IN BREEDING BIRD ASSEMBLAGES OF RIPARIAN FORESTS FROM SW IBERIAN PENINSULA

2. Living on the edge: the effect of habitat quality in breeding bird assemblages of riparian forests from SW Iberian Peninsula

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Keywords: riparian, birds, xeric, quality, richness, abundance, habitat corridor

2.1 Abstract

We conducted a two-year study (2004 and 2006) during breeding season in 2-3 order streams in the left margin of Tagus river basin in Southern Portugal (Mediterranean). The study intended to investigate if riparian habitats supported significantly richer and more abundant assemblages regardless of riparian quality and matrix surrounding. Our aim was also to determine whether we could demonstrate that riparian forests in xeric environments were used as habitat refuges and corridors. A total of 144 points were sampled in the riparian habitat (27 tributaries) and 132 in the matrix, ranging from the following riparian habitat categories: very good to good (2004- 41; 2006- 40), medium (2004- 31; 2006- 36), bad (2004- 26; 2006- 33) and highly degraded (2004- 29; 2006- 35). Point counts were used to census bird assemblages and detection functions were controlled by collecting distances of observation to each bird seem or heard using a rangefinder. Riparian forests supported more diverse and abundant communities independently of matrix and riparian habitat quality; two unique riparian bird species were identified (Nightingale and Cetti's warbler). Common European woodland bird species were more abundant in riparian habitats and were using them as surrogate/refuge. Only 'very bad quality' riparian habitats seem to have a negative impact on species that have been found to prefer riparian environment. Our results support the idea that riparian forests in xeric context act as habitat corridors and refuge for birds.

2.2 Introduction

High species richness and abundance are frequently considered common features in riparian zones (Naiman & Decamps, 1997; Keir *et al*., 2015). The notion that riparian habitats support a rich and abundant fauna in comparison with surrounding non-riparian habitats has important consequences for species conservation, since river networks dissect landscapes and provide a natural framework for conservation planning (Sabo *et al.*, 2005) and connectivity strategies. The habitat function of riparian corridors may be important to re-establish connectivity since some rivers may function as source to spillover effects at landscape level (Didham, 2010; Brudvig *et al.*, 20009; Damschen *et al.*, 2006). In fact, Boulinier *et al.* (2001) pointed out the importance of colonisations and re-colonisations from surrounding landscapes to local community dynamics. We may not be able to detect movement of animals in riparian corridors but the fact that they

often support more diverse and abundant communities at a landscape level may lead them to contribute to maintain connectivity and decrease negative effects of habitat fragmentation.
Most studies with birds have shown higher species richness and abundance in riparian zones (Saunders & Hobs, 1991; Whitaker & Montevechi, 1999; Jobin *et al.*, 2004; Palmer & Bennett, 2006; Cooke & Zack, 2009), tough studies only compared good quality riparian habitats (i.e. habitats with a well-developed riparian gallery) with the matrix surroundings. The higher richness and abundance that riparian areas exhibit has been explained due to mix features of two contrasting environments (aquatic and terrestrial), which causes edge effects (Meffe & Carroll, 1994). However, it seems that this riparian habitat characteristic does not hold in mesic landscapes, where higher or similar diversity has been shown to occur in upland habitats (Murray & Stauffer, 1995; Shirley & Smith, 2005, Seymour & Simmons, 2008), while others have reported the opposite (Palmer & Bennett, 2006). The differences between richness and abundance in mesic and xeric environments have been explained by continuous forests structural features between riparian and upland habitat in Murray & Stauffer (1995) and Whitaker & Montevecchi (1997). Sabo *et al*. (2005) meta-analysis has shown that high species richness in riparian relative to upland habitats is not a general pattern across the globe but the turnover in species pools between these two habitats was a consistent pattern globally in dry and wet climate regardless of the taxonomic group being considered, though being stronger in dry climates.

In Europe riparian bird communities are essentially composed by non-riparian bird specialists (Roché, 2010) and only 4 species of riparian specialists (*Cinclus cinclus, Alcedo atthis, Motacilla cinerea* and *Histrionicus histrionicu*s) have been considered by Buckton & Ormerod (2002) global review. The information of European riparian bird communities is frequently sectorial, mostly limited only to waterbirds (*e.g*. Marchant & Hyde. 1980; Vaughan *et al*., 2007), non-easily accessible (especially from East European countries) (Roché, 2010) and scarce compared to USA and Australia for example. In European Mediterranean areas despite some studies on riparian bird communities (Rabaça, 2004; Godinho *et al*., 2010), a study on the influence of the surroundings on them (Pereira *et al*., 2014) and the development of a bird index associated with river ecological quality (Larsen *et al.*, 2010) no studies have been conducted to investigate if Mediterranean riparian areas support distinctive bird communities. Therefore, studies comparing riparian and adjacent non-riparian habitat are lacking and needed due to the several threats pending on these systems (*e.g*. land use pressures due to agriculture and livestock, climate change). It is expected that in xerofilous Mediterranean matrixes riparian areas support more diverse and abundant species independently of their habitat quality.

In this study we focused on the following research questions: do riparian habitats support different bird assemblages in a xeric environment in the Mediterranean region? Does riparian habitat quality influences bird assemblages? To achieve this, we compared the occurrence of bird species and abundance in riparian and upland habitat (matrix) in different riparian quality forests. The main goals of our study are (1) to compare species richness and abundance between riparian and non-riparian habitats and (2) to evaluate if riparian habitat quality influences riparian bird assemblages. Additionally, we intend to evaluate bird species turnover between riparian and matrix habitats and discuss whether or not we could detect a strictly riparian breeding bird assemblage. The aim of the study was to determine whether we could establish a link between riparian areas as habitat corridors for birds and the development of restoration priority strategies towards connectivity.

2.3 Methods

Study Area

The study was conducted in tributaries of the low section of the left margin of Portuguese section of Tagus river basin (total catchment area of 80 600 km^2- 55 750 km^2 in Spain and 24 850 km^2 in Portugal) (Figure 2.1) in 2-3 order streams (Strahler order rank of rivers) during the breeding seasons of 2004 and 2006. The climate is Mediterranean type with a marked wet season from October to March and a dry season from June to September. Annual rainfall in the study area ranged from 401-500 mm in 2004 (the rainfall was the lowest registered since 1931) and 500-600mm in 2006. The year of 2004 was classified has an extreme dry year, from which the country only fully recovered in 2006, that was classified as dry. The area has dry hot summers (2004 average daily minimum 10.4ºC; 2006 - average daily maxima 21.27ºC) and cool winters (2004 - average daily minimum 10.4ºC; 2006 - average daily minimum 10.81ºC). (IPMA, 2004; IPMA, 2006).

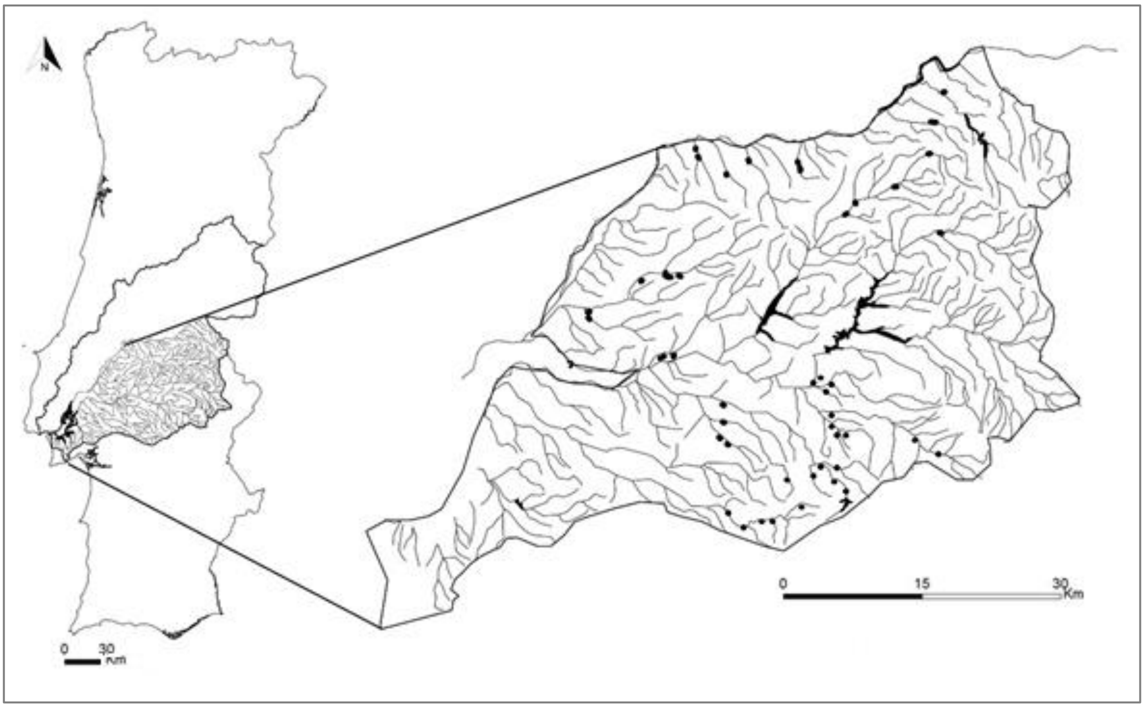

Figure 2.1 – Location of the study area (Portuguese section of Tagus river basin-left margin) and distribution of the sampling units where the river stretches were surveyed. *Black dots* identify the location of the set of six sampling points located 500m apart from each other (three in the riparian habitat and three pairs in the matrix habitat).

Catchment topography varies from medium slopes to alluvial valleys in the lower parts of the area, closer to the Tagus River. Tributaries range from all-year running to temporary reaches. Greater habitat heterogeneity occurs in less populated areas mainly in the tributaries, since the main water course is highly impacted with dams for hydropower generation. Woody riparian vegetation comprises Common alder (*Alnus glutinosa* L., Gaertner), Sage-leaved willow (*Salix salvifolia* Brot. ssp. *australis* Franco), Rusty sallow (*Salix atrocinerea* L.), Brittle willow (*Salix neotricha* Goerz*)*, Oleander (*Nerium olean*der L.), Narrow-leafed ash (*Fraxinus angustifolia* Vahl) and Alder buckthorn **(***Frangula alnus)*. Stands of African tamarisk (*Tamarix africana* Poiret) and Oleander occur in the dry interior parts of the river basin. In the riparian understory occur bramble (*Rubus ulmifolius* Schott), wild roses (*Rosa* spp.), Common ivy (*Hedera helix* Nyman) and Common smilax (*Smilax aspera* L.).

Most frequent landscape matrices in riparian areas include natural Mediterranean cork-oak woodland (*Quercus suber* L.) or Holm oaks (*Quercus rotundifolia* L.), Olive groves (*Olea europea*

L.), some *Eucalyptus* sp and *Pinus* sp. plantations. Kermes oak (*Quercus coccifera* L.), Hawthorn (*Crataegus monogyna* Jacq), Gorces (*Ulex* spp.) and Rock roses (*Cistus* spp.) occur frequently in the shrub layers. Common land uses are cattle raising (cows, sheep and pigs) in association with the cork-oak and holm oak woodlands. *Eucalyptus* plantations occur particularly in the Northern part of the area. In the middle section (close to the Tagus River) rice, tomato and corn are cultivated, which contrasts with the southern part of the area that has cereal production. High urbanization occurs only close to large villages. Impacts in the study area include diffuse pollution from agriculture (particularly from rice, tomato and corn plantations and cattle raising), physical disturbance (riparian forest removal, bank reinforcement and channelization) and reduced longitudinal connectivity (due to water abstraction for agriculture purposes). In less populated areas the riparian galleries are well developed and continuous, which contrasts with lower sections closer to Tagus River where they are affected by agriculture, since water availability is high.

Study sites

Site selection was made to have good spatial coverage of the area. Data were collected from a total of 55 study sites (Figure 2.1) located in the 27 tributaries: Agolada (2), Alecrim (1), Almânsor (4), Arraiolos (2), Balancho (2), Barrosas (2), Cabido (2), Chouto (1), Divor (3), Erra (2), Fanica (2), Fernando (2), Figueiró (1), Fontainhas (3), Foz (2), Freixo (2), Grou (2), Lamarosa (2), Mar (2), Muge (6), Nisa (1), Pigueira (1), Seda (2), Sor (3), Tera (3), Torto (1), Ulme (3). Whenever possible, three sampling points were conducted in the riparian habitat and in the adjacent matrix. Matrix points were coupled parallel to the riparian point at a distance of 500m. Since according to Bibby *et al.* (2005) 100m is the limit distance to independent bird sampling, points in each habitat were 500m apart from each other.

A gradient ranging from pristine to highly disturbed riparian galleries was established using the QBR index (Munné *et al*., 2003). This index has 5 categories: ≥95 very good quality (pristine), 75-90 good quality (little disturbance), 55-70 medium disturbed (satisfactory, with significant intervention), 30-50 bad quality (not satisfactory, with marked anthropogenic modification), less than 25 highly degraded (marked degradation). Since we have only found 3 points higher than 95, we only used a category for very good to good quality. A total of 144 points were sampled in the riparian habitat and 132 in the matrix, ranging from the following riparian habitat categories:

very good to good (2004- 41; 2006- 40), medium (2004- 31; 2006- 36), bad (2004- 26; 2006- 33) and highly degraded (2004- 29; 2006- 35).

Bird surveys

Point counts were used to census bird assemblages (*e.g*. Bibby *et al*., 2005). Birds were surveyed only once in the breeding seasons of 2004 and 2006, between April and mid-June, the standardized period for surveying birds in the region. During each visit all birds seen or heard were recorded during 5 min point counts. Sampling started 30 min after sunrise and was not conducted during rain or when winds exceeded 20km/h since poor weather conditions reduces the activity and detectability of birds (following Bibby *et al*., 2005). Surveys were carried out by two experienced observers with similar skills of aural and visual bird recording and detection functions were controlled by collecting distances of observation to each bird seem or heard using a rangefinder (Leica Disto 5a). A variable limit of point count detection was used so that birds that were only in the riparian area were recorded in an attempt to restrict the observations to the habitat being sampled (Whitaker & Montevecchi, 1997). The limit of detection in riparian areas was the length of the riparian area and 50m for each side of the river stretch surveyed. In the matrix a radius of 100m was used. These detection distances were consistent with both observer histograms of the distance of detection in the two types of habitats.

Habitat features

Riparian area quality was assessed using the following QBR parameters: total vegetation cover; vegetation cover structure; cover quality; river channel alterations (Munné *et al.*, 2003). The total width of the riparian area was used to collect the data. Recorded features were collected in one attempt to characterize the gradient of degradation and river habitat condition. Matrix habitat categories were recorded and included in the following categories: ‘montados’ (woodland and holm oaks forests), olive groves, *Eucalyptus* forest, pine forests, rice fields, irrigated crops and pastures. Sand mining areas were also recorded.

Data analysis

Bird abundances were standardized for analysis since birds were surveyed on different scales. In the matrix counts were truncated at 100m based on the assumption that in woodland areas the listening range for songbirds does not exceed a radius of 100m (Decamps *et al.*, 1987). Detected birds in each habitat were divided by the corresponding area of the point count. Birds counted in the matrix were divided by the area of a circle with 100m radius. In the riparian habitat birds were divided by twice (to account for up and downstream detection) the width of the riparian habitat multiplied 50m (Figure 2.2), since this was the consistent detection limit of the two observers in this type of habitat. To analyse if detection functions of observers were different Mann-Whitney U test was used.

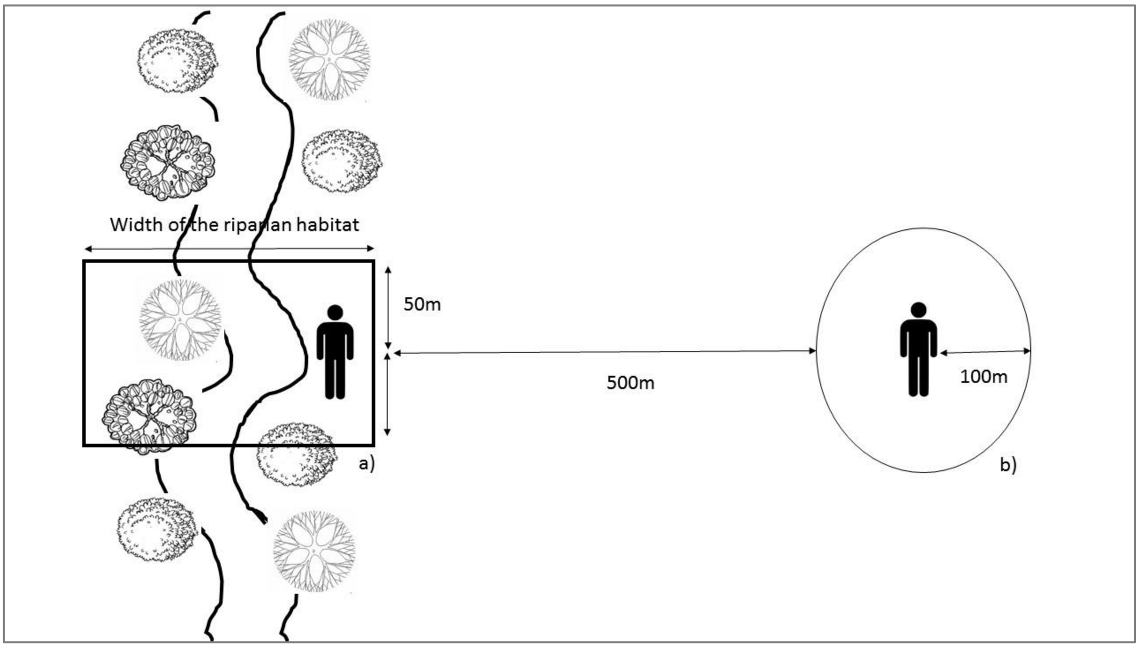

Figure 2.2 - Scheme showing an example of one point of the sampling site with the corresponding surveyed areas in the riparian habitat (a) and the matrix (b). In each site three pairs (a) and (b) were established.

To analyse if species richness and abundance in riparian habitat significantly differed from the matrix, Wilcoxon matched pair tests were conducted. We performed analysis of similarity (ANOSIM) using the PRIMER software package (Clarke & Gorley, 2001) to detect differences in species composition between riparian and matrix habitats. Similarity Percentage Analysis – SIMPER - was used to identify the bird species that contribute most to the similarities within sites (riparian and matrix) and the dissimilarities between sites based on contributions of bird species to the Bray-Curtis similarity matrix. Variables were not transformed to give relevance to the most abundant species, and we did not excluded species not present in less than 5% of the sampling points because we intended to focus in the most common species. We also performed an ordination of bird assemblages at each site using multidimensional scaling (MDS) based on the Bray-Curtis similarity matrix. Species abundance and composition at each type of surveyed habitat were performed using SIMPER and Permutational Multivariate Analysis of Variance – PERMANOVA - procedures to analyse if the type of matrix habitat had influence on results and to study if the decay of riparian habitat quality had influence on identifying a riparian bird community. Kruskall-Wallis tests were performed to explore if the species identified by the SIMPER routines had statistical significant different abundances in the several riparian habitat quality categories surveyed.

2.4 Results

Differences between observers/years

Observers detection function (distance detection functions) were statistically different from each other in the riparian habitat ($z=16.99$; $p<0,001$) and in the matrix ($z=11.39$; $p<0.001$), so data from each year was analysed separately. Though the limit of detection of both observers was the same in both habitats: 50m for riparian areas and 200m for matrix habitats, the majority of records in the matrix occurred within 100m.

Bird assemblages

A total of 75 bird species was detected in the study area: 66 species were recorded in 2004 and 65 species in 2006 (Table 2.1 in Appendix I). In both years (2004 and 2006) species richness was higher in the matrix than in the river corridor: In the riparian zone 49 and 42 species were recorded (26 and 33 species un-recorded), whereas 55 and 58 species were detected in the matrix (20 and 17 species unrecorded) respectively in 2004 and 2006. Five species were recorded only in the riparian area, though only two are typical of aquatic ecosystems (Grey Heron *Ardea cinerea,* Kentish Plover *Charadrius alexandrinus*). Ten species were detected only in the matrix. Three typical aquatic species (Black winged stilt *Himantopus himantopus*, Night heron *Nycticorax nycticorax* and Spoonbill *Platalea leucorodia*) were recorded in the matrix in rice fields.

Bird richness in riparian forests was significantly higher when compared to the matrix pair in both years (2004: z=9.38; p<0.001; n=126; 2006: z=9.63; p<0.001; n=142). The differences in richness between the riparian habitat and the matrix were respectively: 2004 – 10.4 species (±6.1; range 0-18) and 1.2 species (±0.7; range 0-3); 2006 – 9.7 species (±7.4; range 1-60) and 1.5 (±1.5; range 0-15). No significant differences were detected in the richness between years in both habitats (Mann-Whitney U test: matrix z=1.89; p>0.05; river z=-1.49, p>0.05). Bird abundances were significantly different between riparian habitat and the matrix (2004: z=9.73; p<0.001, n=126; 2006: z=9.63; p<0.001, n=142). The relative abundance was always greater in the riparian habitat (2004 – (30.9 ±2.4)*10^{-4}; range 1-190; 2006 – (10.9±0.8)*10^{-4} ; range 1-85) than in the matrix (2006 – (2.1±2.3)*10^{-4}; range 0-18; 2006 – (1.9±0.2)*10^{-4}; range 0-99).

The bird assemblages differed significantly between riparian habitat and the matrix in both years (ANOSIM: 2004 *R*=0.38, *p*<0.001; 2006 *R*=0.47, *p*<0.001) independently of the quality of the riparian habitat and type of matrix surrounding it. A SIMPER analysis (Table 2.2) revealed that 6 and 4 species (for 2004 and 2006 respectively) contributed approximately to 70% of the similarity occurring in riparian habitats, whereas for the matrix 6 and 7 species were identified. The major contributors (up to 50%) for similarity within riparian habitats were: 2004 – Cetti's warbler *Cettia cetti* (22.3%), Nightingale *Luscinia megarhynchos* (19.2%) and Blackbird *Turdus merula* (9.5%); 2006 – Nightingale (36.8%), Cetti's warbler (15.5%) and Eurasian blackcap *Sylvia atricapilla* (11.2%). The major contributors for similarity within the matrix were: 2004 – Fan-tailed warbler *Cisticola juncidis* (34.7%), Stonechat *Saxicola torquata* (10.8%), Corn bunting *Miliaria calandra* (8.6%); 2006 – Serin *Serinus serinus* (19.4%), Corn bunting (18.5%), Common chaffinch *Fringilla coelebs* (13.6%). However, there were species common to both habitats that appear in the top 70% contributors: 2004 – Sardinian warbler *Sylvia melanocephala* (riparian habitat 9.1%, matrix 8.51%) and Serin (riparian habitat 3.6%, matrix 5.6%; 2006 – Serin (riparian habitat 4.7%, matrix 19.4%), Greenfinch *Carduelis chloris* (riparian habitat 5.2%, matrix 5.7%) and Blackbird *Turdus merula* (riparian habitat 7.8%, matrix 5.3%). Thirteen species from 2004 and and twelve species from 2006 accounted for 70% of dissimilarity between bird assemblages of riparian and matrix habitats. The greatest contributors common to both years were: Cetti's warbler (2004: 10.1%; 2006: 9.43%); Nightingale (2004: 9.8%; 2006: 14.9%); Blackbird (2004: 5.9%; 2006: 6.7%). In 2004 other important contributors were Common waxbill *Estrilda astrild* (6.6%); Sardinian warbler

(6.3%); Iberian Chiffchaff *Phyllocopus ibericus* (5.5%), Blue tit *Cyanistes caeruleus* (5.3%). In 2006 other contributors were: Greenfinch (5.9%) and Wren *Troglodytes troglodytes* (5.8%). These species are the ones that exhibit larger contrasts in relative abundance between habitat types (Table 2.1 in Appendix II).

Table 2.2 – Results of SIMPER routine showing species contributing up to 70% to riparian and matrix similarity and species contributing up to 70% to dissimilarity between riparian and non-riparian habitats. Abbreviations in AppendixIV3

	2004		2006	
Habitat	Species	Contribution%	Species	Contribution%
Riparian	Cet cet	22.33	Lus meg	35.8
	Lus meg	19.20	Cet cet	15.51
	Tur mer	9.47	Syl atr	11.23
	Syl mel	9.14	Tur mer	7.8
	Phy ibe	7.52		
	Syl atr	6.45		
Matrix	Cis jun	34.74	Ser ser	19.38
	Sax tor	10.78	Mil cal	18.51
	Mil cal	8.56	Fri coe	13.57
	Syl mel	8.51	Cya cae	6.54
	Fri coe	7.66	Car chl	5.73
	Cya cae	6.64	Tur mer	5.27
	Ser ser	5.88	Cis jun	4.79
Dissimilarity between riparian and matrix	Cet cet	10.08	Lus meg	14.93
	Lus meg	9.83	Cet cet	9.35
	Est ast	6.67	Syl atr	7.53
	Syl mel	6.30	Tur mer	6.74
	Tur mer	5.89	Car chl	5.9
	Phy ibe	5.50	Tro tro	5.84
	Cya cae	5.31	Ser ser	5.05
	Syl atr	4.65	Cya cae	3.68
	Par cae	4.27	Phy ibe	3.50
	Ser ser	4.02	Hip pol	3.48
	Cis jun	3.98	Fri coe	3.12
	Hip pol	3.12		

Two (one for 2004 and another for 2006 data) MDS ordination of sampling stations based on the composition of bird assemblages displayed contrast between riparian and matrix habitats (Figure 2.3) although moderate fit of the data (2004 stress=0.17; 2006 stress=0.18). The matrix and riparian habitats show similar dispersal among the space.

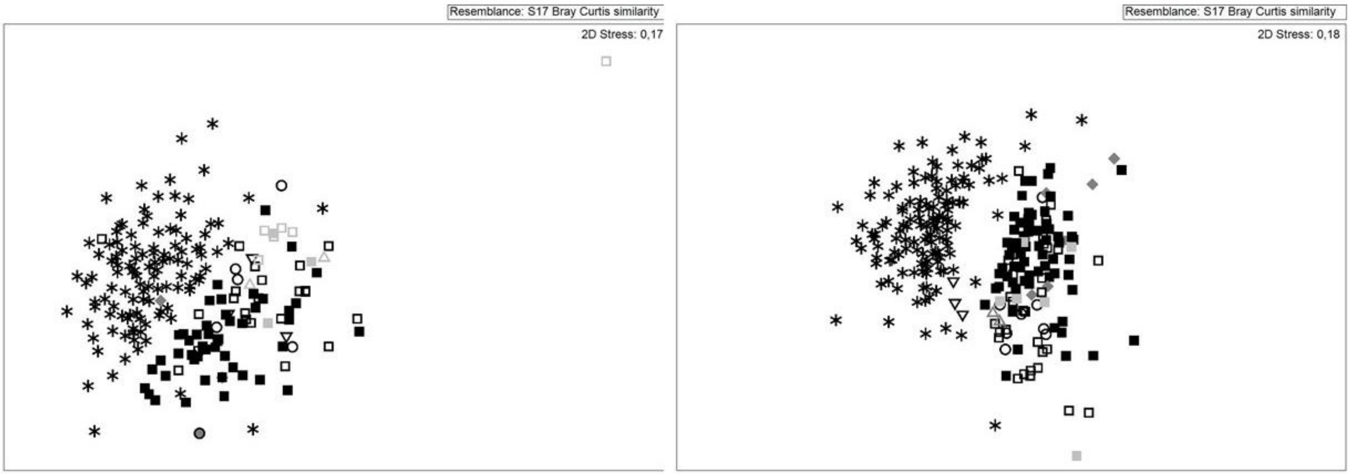

Figure 2.3 – Ordination of bird assemblages occurring at the sites in Tagus river basin (2004 stress=0.17; 2006 stress=1.18). The different habitats are displayed: Riparian forest (*);Montado (■); Pine (grey ■); Pasture (□); Olive groves (○); Eucaliptus forest (grey ◆) sand extraction (grey outline Δ); Rice fields (grey outline □); Irrigated crops (▽).

Landscape influence

The results of the pairwise tests of ANOSIM between the different types of habitat pairs (Table 2.3 in appendix 2) surveyed in both years revealed that although differences between riparian habitats and the matrix are small they are significant (2004: *Global R*= 0.356, p<0.001; 2006: *Global R*=0.436, p<0.001). In 2004, all pairwise tests between the riparian habitat (independently of their quality) and the different matrix habitats were significantly different (p<0.001) except for the pair *Eucalyptu*s forest and riparian forest; whereas in 2006 the pairs riparian and sand extractions and riparian and irrigated crops did not show significant differences (p>0.05). In contrast with the pairs riparian and 'montado', riparian and olive groves, riparian and *Eucalyptus* forest and riparian and pine forest that did exhibit significant differences (p<0,001). Table 2.3 (in appendix 2) summarizes the results of the SIMPER routine where we can identify the species that contribute up to 70% or more to each type of habitat and respective relative abundances. Results are roughly consistent between years and species that are characteristic of riparian areas only, are Nightingale and Cetti's warbler, since other species (*e.g.* Eurasian blackcap, Sardinian warbler, Blackbird) also occur in other habitats in different relative abundances.

Riparian habitat quality influence

The PERMANOVA results show that irrespective of the riparian habitat quality the comparisons between the pairs riparian habitat and matrix habitats with degrees of freedom higher than 10 were always significantly different in both years (Table 2.4), except for pasture (in 2004). Moreover, in very bad quality riparian habitats the following pairs exhibited significant differences: riparian habitat and irrigated crops (in 2004); riparian habitat and ‘montado’ (in 2006) and riparian habitat and pasture (in 2006).

Table 2.4 – Summary of PERMANOVA results comparing riparian habitats and the different matrix habitats

		2004			2006		
		p-value	df	average sim	p-value	df	average sim
QBR Good	Quercus - Montado	p<0.05	30	1.35	p<0.05	41	2.11
	Pine	-			-		
	Pasture	p<0.05	12	1.25	ns	5	3.68
	Eucalitus	ns	6	0.76	ns	16	1.380
	Olive groves	ns	2	0	ns	4	3.00
	Rice fields	-			-		
	Irrigated crops	-			-		
	Sand extraction	ns	2	0	ns	4	6.94
QBR Medium	Quercus - Montado	p<0.05	29	2.19	p<0.05	36	2.01
	Pine	-			-	0	
	Pasture	p<0.05	18	2.44	p<0.05	23	1.43
	Eucaliptus	-			-		
	Olive groves	-			-		
	Rice fields	-			-		
	Irrigated crops	ns	1	2.81	-		
QBR Bad	Quercus - Montado	p<0.05	25	1.37	p<0.05	35	1.389
	Pine	-			-		
	Pasture	-			p<0.05	11	1.39
	Eucaliptus	-			-		
	Olive groves	-			-		
	Rice fields	ns	1		ns	4	3.07
	Irrigated crops	ns	2	1.34	ns	4	10.53
QBR Very Bad	Quercus - Montado	ns	6	0.7	p<0.05	17	1.36
	Pine	ns	8	2.69	ns	4	1.34
	Pasture	ns	15	0.96	p<0.05	17	1.26
	Olive groves	ns	5	0.2	ns	6	1.89
	Irrigated crops	p<0.05	15	0.76	ns	4	1.40

The 2004 SIMPER routine identified nine species that contributed to 70% similarity within groups of different QBR categories (Nightingale, Eurasian blackcap, Cetti's warbler, Iberian chiffchaff, Melodious warbler *Hippolais polyglotta,* Kingfisher, Blackbird, Serin and Common waxbill), posterior Kruskal-Wallis tests revealed significant differences between abundances of the following bird species in 2004: Nightingale (H(3, 126)=8.38; p<0.05); Iberian chiffchaff (H (3, 126)= 8.14; p<0.05) and Eurasian blackcap (H(3,126)=17.55; p<0.001) for the different QBR categories. Multiple comparisons tests only detected significant relative abundance differences in Eurasian blackcap between good and very bad riparian habitats (p=0.02) and marginally significant differences between good and very bad habitats for Nightingale (p=0.07). For 2006, SIMPER routine identified six species that contributed to similarity within groups of different QBR categories (Cetti's warbler, Eurasian blackcap, Nightingale, Serin, Wren and Blackbird), however posterior Kruskal-Wallis tests did not reveal significant differences between abundances of any of the bird species in 2006.

2.5 Discussion

Riparian forests exhibited significantly richer and more abundant assemblages than their matrix pairs in both years. This result alone is well known and consistent with findings from other studies in other regions (Saunders & Hobs, 1991; Whitaker & Montevechi, 1999; Jobin *et al.*, 2004; Palmer & Bennett, 2006; Cooke & Zack, 2009). However, we did find that these results are also consistent independently of riparian habitat quality and matrix surrounding. Results are supported by two years of sampling with different observers, so same conclusions are drown in different years with slight species composition differences.

Bird assemblages

Despite the fact that riparian habitats supported richer and more abundant bird assemblages than the matrix we expected to detect a marked contrast between these habitats and the xeric landscape (Sabo *et al*., 2005) which was not detected. Cetti's Warbler and Nightingale were the only species unique to riparian forests, while Blackbird and Blackcap although most abundant in riparian forest they also use the matrix and are common forest species that occur even in city parks. These findings can be explained by bird mobility characteristics, and intrinsic (*e.g*. heterogeneity) and extrinsic (*e.g*. surrounding matrix type) habitat characteristics (Pereira *et al.*, 2014).

The microclimatic characteristics of riparian galleries are well known in Mediterranean habitats where they are known to increase humidity and reduce temperature due to evapotranspiration while also providing shade (Sabo *et al*., 2008) The fact that we have found typically woodland common European birds species (*e.g*. blackbird, tits, blackcap) in riparian forest in higher numbers compared to the matrix can be explained by higher moister conditions that are consistent with their ecological range (Pereira *et al.*, 2014), moreover riparian forest are frequently the single forested habitat in open areas of Mediterranean type landscape, which may represent the only breeding and feeding site available to several bird species as was reported in other regions with similar characteristics (Deschênes *et al*., 2003; Strong & Bock, 1990; Pereira *et al.*, 2014; Nimmo *et al*., 2016).

Matrix characteristic species are also common bird species associated with agriculture farmland (Corn Bunting, Fan-tailed Warbler, Chaffinch, Serin and Stonechat) some of which also occur in urbanized environments. These results are consistent with the findings in Pereira *et al*. (2014) for the same region, which found that Corn bunting and Fan-tailed warbler occur in matrix habitats in open areas and sparse ’montado’ respectively. While Serin, Chaffinch and Stonechat are characteristic of farmland areas they were detected also associated to riparian galleries in both studies nevertheless with higher relative abundance in matrix habitats. Mobile animals such as birds may take advantage of seasonally favorable microclimates, surface water or other riparian conditions and resources, despite significant dependence on upland habitats (Sabo *et al*., 2005; Sabo *et al*., 2008).

Influence of the surroundings

We found consistent differences, in species composition and structure of abundances, between riparian forest and the several matrix habitats surveyed. In 2004 all matrix habitats (except *Eucalyptus* forest) were significantly different from the surroundings, and in 2006 only sand extraction and irrigated crops did not show a significant difference when compared to the riparian forest. We are aware that replication of the several matrix types is far from ideal; nevertheless

’montado’ forest, olive groves and pasture, which constitute the large proportion of surrounding landscape type in the river basin were replicated properly. From the analysis of Table 2.3 (Appendix II) we can see that the relative abundance of birds is always higher in the riparian forest (except for Wood pigeon *Columba palumbus* in irrigated crops, but this is probably due to the flocking behavior of the species) and that several common woodland species that contribute up to 70% to the riparian habitat also appear in other matrix habitats (*e.g*. Blackcap and Blackbird) indicating that birds are using the riparian gallery as a surrogate/refuge habitat as it has been reported in xeric environments (Simmons & Allan, 2002; Palmer et Bennet, 2006; Seymour & Simmons, 2008; Pereira *et al*., 2014). We could not identify an exclusive riparian assemblage since common bird species also occur in other typologies of matrix habitats, nevertheless we can say that Cetti’s Warbler and Nightingale are unique riparian birds in Mediterranean climates (Godinho *et al*., 2010; Larsen *et al*., 2010).

Influence of riparian habitat quality

Habitat structure is a major determinant of bird community composition and many studies have demonstrated a link between riparian vegetation and birds in riparian zones, suggesting that changes in bird community composition can be predicted from changes in habitat structure (Rottenborn, 1999). Biodiversity tends to be reduced due to habitat fragmentation, which normally occurs when human activities reduce contiguous forest patches into smaller isolated remains. Several studies demonstrated that fragmented riparian forests have conservation value for riparian bird species (Bentley & Caterall, 1997; Skagen *et al*., 1998; Palmer & Bennet, 2006; Seymour & Simmons, 2008). In this study we demonstrate that independently of riparian habitat quality, riparian forests still play an important role at landscape level as surrogate habitat for bird species. We compared matrix habitat types with different riparian habitat quality and found that irrespective of the latter, the riparian forest plays a role in supporting bird assemblages in xeric landscapes. Results reveal a negative tendency towards riparian degradation, with some riparian species showing a significant decrease when we compare good and very bad quality riparian forests. Hughes *et al.* (2009), Godinho *et al*. (2010) and Larsen *et al*. (2010) have demonstrated the usefulness of bird assemblages as predictors of riparian degradation. In fact, Hughes *et al*. (2009) have demonstrated that large, mobile organisms such as birds provided reliable links between organism structure and function, environmental factors and physical disturbance of the channel, bankside and wider river corridor.

Several factors have been reported to affect bird diversity and abundance in riparian forests: dominant tree species (Strong & Bock, 1990, Powell & Steidl, 2000), riparian width (Hagar, 1999; Pearson & Manuwal, 2001; Scott *et al*., 2003; Cooke & Zack, 2009), height (Cooke & Zack, 2009), area (Groom & Grubb Jr, 2002), adjacent upland vegetation (Strong & Bock, 1990), stream geomorphology and insect prey (Iwata *et al*., 2003), hydrological characteristics (Royal *et al.*, 2013). While some researchers found that plant species composition (diversity and heterogeneity) affect bird diversity and abundance (Sanders & Hobs 1991; Powell & Steidl, 2000; Scott *et al.*, 2003) others did not find any relationship between riparian bird communities and plant species richness or dominance of non-native plants, provided that vegetation community retains sufficient structural diversity (*e.g.* Fleishman *et al.*, 2003). In the Guadiana river basin Godinho *et al.* (2010) reported that both rivers with high vegetation cover of riparian species and rivers dominated by rock and scarce emergent rooted vegetation were good predictors of passerines and aquatic/riparian species.

Conservation implications

Our results did not show that riparian forests in Mediterranean-type habitats support a unique assemblage of birds (only two bird species unique of these habitats) in the breeding season, although they support a richer and more abundant bird assemblage compared to other matrix typologies. This lack of uniqueness in breeding bird communities associated to riparian forests in southwestern Iberian Peninsula is not surprising because Mediterranean ecosystems have been submitted for centuries to huge modifications imposed by human use (*e.g*. Blondel & Aronson, 1999; Hughes *et al*., 2009) which have altered natural habitats to a great extent, including forest fragmentation, reclamation of flooding plains for agriculture and the destruction of riparian forests for wood and charcoal demands. Therefore, current landscapes in SW Iberia are a mixture of natural and ruderal plant elements due to a millenary history of human disturbances. The reduction of forested area and disappearance of several forest types, promoted habitat generalism among forest bird assemblages (*e.g*. Mökkönen & Welsh, 1994) and because breeding bird communities in forested temperate and Mediterranean areas are dominated by resident birds (see Blondel & Farré, 1988) these species might have taken advantage in the use of semi-natural areas. Our results suggest that common European woodland birds (i.e. widespread species which are often abundant) use riparian areas as refuge habitats in xeric landscapes, which support also the recent findings of Nimmo *et al.* (2016), thus are using them as habitat corridors, which is very important considering the general decline of European birds (Inger *et al*. 2015). Also, climate change is estimated to have impact on common resident birds (Sekercioglu *et al.,* 2008; Nimo *et al*., 2016).

River networks connect landscapes and provide natural framework for conservation planning (Sabo *et al*., 2005). Riparian areas are ideally suited to form the basis of linked wildlife habitat networks because they form a hierarchy of natural dendritic corridors throughout the landscape (Knopf & Samson, 1994; Perry *et al.*, 2011), are used by most forest-dependent species (*e.g.* Jobin *et al*., 2004), act as buffers to protect water quality and aquatic ecosystems (Naiman & Decamps, 1997; Cushman *et al.*, 2013) and potentially serve as refuges (Palmer & Bennet, 2006; this study).

It is estimated that nearly 70% of vertebrate species in a region will use riparian areas in some significant way during their life cycle (Raedeke, 1989). In fact, our study shows that some farmland species (Serin, Chaffinch and Stonechat) were also detected in riparian galleries nevertheless with higher relative abundance in matrix habitats. Though they are common birds, these observations support the idea that reserves designed to protect upland habitats per se, may fail to protect mobile taxa dependent on multiple habitats, unless some combination of riparian and upland habitats is considered together in a more complex plan (Sabo *et al*., 2005).

The European Water Framework Directive (WFD) (Council Directive 2000/60/EC), Habitats Directive (HD – Council Directive 92/43/EEC) and Birds Directive (BD – Council Directive 2009/147/EC) as well as EU2020 Biodiversity Strategy set goals for restoring degraded ecosystem so that ecosystem services are maintained. The restoration of freshwater ecosystem is mainly driven by WFD and its quality indicators do not use birds, however, an articulation of the several Directives to set priority strategies to achieve multipurpose goals should be developed so that riparian restoration could also provide a natural framework to link Natura 2000 sites and WFD demands. The inclusion of birds as an element to be monitored for conservation and linked with WFD biological elements remains to be analysed and has been explored by Hughes *et al*. (2009) and Larsen *et al.* (2010).

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Chapter 3

ECOLOGICAL ASSESSMENT OF AN INTERMITTENT MEDITERRANEAN RIVER USING COMMUNITY STRUTURE AND FUNCTION: EVALUATING THE ROLE OF DIFFERENT ORGANISM GROUPS

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3. Ecological assessment of an intermittent Mediterranean river using community structure and function: evaluating the role of different organism groups

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Key words: Mediterranean rivers, fish, birds, macroinvertebrates, macrophytes, structure, function, bioassessment

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3.1 Summary

1. Reliable lotic ecological monitoring requires knowledge of river typology, environmental factors, the effect of stressors known here as “pressures” and appropriate indicators of anthropogenically induced change. We sampled benthic macroinvertebrate, fish, bird and macrophyte communities along an intermittent Mediterranean river and analysed community structure (relative abundance) and function (metrics) relative to environmental and pressure gradients in order to identify suitable indicator group(s) for future monitoring and mitigation programmes.

2. Principal Components Analysis (PCA) revealed that scale-dependent longitudinal differences in valley form separated narrower higher lying sites and tributaries with good quality habitats from more open degraded sites lower down the river continuum on a small floodplain and large scale pressures describing changes in land use related to agriculture with associated physical bankside and channel impacts.

3. Forward selection of variables in Redundancy Analysis (RDA) showed that reach scale environmental variables were selected more frequently than pressure variables for each organism group. Altitude and pH were highly redundant within and between groups, indicating essentially longitudinal structural and functional distribution patterns. Redundancy was far lower between selected pressure variables, but single or no pressure variables were retained for some organism groups indicating poor association of functional data, in particular, with the identified pressures. All RDA results indicated a longitudinal pH gradient, highlighting the combined effect of multiple environmental and pressure based mechanisms on organism groups.

4. Large, mobile organisms such as fish and birds provided a reliable link between organism structure and function, environmental factors and physical disturbance to the channel, bankside and wider river corridor. Benthic macroinvertebrate and macrophyte structural data revealed distribution patterns in relation to water velocity, a key parameter for developing appropriate compensation measures.

5. Results clearly show the importance of assessing patterns of both functional and structural change across multiple organism groups in order to identify typologically appropriate links with complex environmental and pressure gradients and develop and implement appropriate monitoring systems.

3.2 Introduction

Ecological assessment methods of lotic systems have tended to focus on single groups, in particular benthic macroinvertebrates, fish, diatoms and macrophytes (Barbour *et al.*, 1999; Furse *et al.*, 2006). More recently, the wider stream environment has been successfully assessed using birds (Bryce, Hughes & Kaufmann, 2002; Vaughn, Noble & Ormerod, 2007b). Given the hierarchical, multivariate complexity of river systems, it could be argued that single group monitoring methods have limited applicability and that multiple organism group approaches provide a more comprehensive ecological image of river health. The multiple organism group approach has gained considerable momentum in recent years, both in the USA (Barbour *et al.*, 1999) and Europe, in particular since the implementation of the EU Water Framework Directive (Bryce *et al.*, 2002; Hering, Johnson & Buffagni, 2006b; Hering *et al.*, 2006c; Johnson *et al.*, 2006a; Johnson & Hering, 2009; O'Connor, Walls & Hughes, 2000). This approach assumes that different life history strategies of different communities will respond (in a measurable or quantifiable way through structural or functional changes in the community assemblage) to different types of stressors, providing complementary and comprehensive information on ecological status and pressures affecting the system (Hering *et al.*, 2006c). Arguments against the multiple organism group approach include experimental design flaws, high costs, complicated logistics concerning implementation, potentially high levels of redundancy (i.e. fewer or even a single group can potentially provide relevant information on ecological status for management needs) and often inconclusive results (Resh, 2008).

Lotic biomonitoring programmes must be appropriate to river typology and based on biological indicators that respond to pressures in a way that is distinct from natural variation (Hering *et al.*, 2006c). This means that suites of ecological indicators can vary between river types within a given ecoregion. For example, Johnson *et al.* (2006a) found that fish, macroinvertebrates, macrophytes and benthic diatoms exhibited different degrees of response to stress gradients in European mountain and lowland streams. Mediterranean river typology is characterised by a predictable annual cycle of flood and drought that vary in intensity according to levels and duration of annual and interannual rainfall (Bêche & Resh, 2007a; Bêche & Resh, 2007b; Bonada, Rieradevall & Prat, 2007; Gasith & Resh, 1999; Pires *et al.*, 2004). This regime acts as an extremely strong environmental filter, shaping biological community traits (Bonada *et al.*, 2005; Poff, 1997; Statzner, Dolédec & Hugueny, 2004), a potential source of environmental covariance concerning biotic reactions to anthropogenic pressures. Iberia's

Mediterranean rivers have a long history of human intervention including intensive agriculture and forestry, damming, abstraction and urbanisation, resulting in habitat fragmentation, soil erosion, reduced connectivity and disruption of natural flow regimes. These impacts compromise lotic structure and function in systems already subject to a harsh natural seasonal cycle of drought and flood (Aguiar & Ferreira, 2005; Hooke, 2006; Hughes, Ferreira & Cortes, 2008).

The incumbent Portuguese Government has authorised completion of a partially constructed dam on the Odelouca River by 2010 in order to augment and improve water supply in the Algarve region, but has demanded environmental mitigation and compensation measures to offset impacts caused during and after construction. The Odelouca, an intermittent Mediterranean river in the Algarve region of southern Portugal has been subject to human intervention along much of its catchment, but has high conservation value due to relatively intact and floristically unique riparian galleries found along stretches of the river corridor and the presence of two critically endangered endemic fish species (Pires *et al.*, 2004), the Iberian Chub *Squalius aradensis* (Coelho *et al.*, 1998) and the Iberian nase *Iberochondrostoma almacai* (Coelho, Mesquita & Collares-Pereira, 2005).

This study aims to assess the structural and functional response of four organism groups (benthic macroinvertebrates, fish, birds and macrophytes) to environmental factors and pressures in an intermittent Mediterranean system and assess their suitability for future monitoring programmes and mitigation measures. In this study we have divided variables into (i) environmental variables, which essentially describe natural features such as valley form, altitude, the presence of depositional bars and pH and (ii) pressure variables describing types of impact such as changes in land use (agriculture, urbanisation) bankside disturbance and discharge into the watercourse. We also adopt the definition of structural (taxonomic composition and abundance) and functional composition (groups of metrics covering several properties of the communities under study) described by Feld & Hering, 2007. Although metrics (i.e. functional data) are more commonly used for expressing ecological condition, changes in relative abundance can also provide important information on ecosystem patterns and may even contribute to the development of new metrics. A study of the effect of environmental stress on benthic macroinvertebrate assemblages in Central European lowland rivers found that levels of explained variance were higher for taxa (structure) than for metrics (function) (Feld & Hering, 2007). We have investigated the relation of these organism groups to natural and stressor gradients in an intermittent Mediterranean river in order to compare the results of structural and functional data and assess candidate organism groups that best describe human induced change within the study area.

3.3 Methods

Study Area

The Odelouca river (511 km^2), a sub-catchment of the Arade basin (987 km^2) is a medium-sized, low-gradient, lowland stream running through predominantly schistose areas characteristic of southern Portugal (Fig. 3.1). The climate is typically Mediterranean; mean monthly precipitation levels (Bravur Reservoir, 15 km west of the study area, hydrological year 2004-2005), exemplify the seasonal rainfall pattern with a range of 59 mm – 102 mm precipitation per month (mean value) from October 2004 to April 2005 and 2 mm – 38mm mm precipitation per month (mean value) from May to September 2005. Precipitation patterns result in a relatively slow running river subject to “flashy” spates in winter that dries to form unconnected, temporary pools (intermittent reaches *sensu* Gasith and Resh 1999) in the river bed during the summer.

Catchment topography varies from narrow steep sided valley walls to restricted meander valleys and small floodplains in the lower reaches. Temporary side channels, backwaters and dense riparian galleries occur in less disturbed mid sections and tributaries, which will be submerged following dam completion. Dominant woody riparian plants comprise *Alnus glutinosa* (L.) Gaertner, *Salix salviifolia* Brot. spp. *australis* Franco, *Nerium oleander* L, and *Fraxinus angustifolia* Vahl.. Stands of *Tamarix africana* Poiret and *Nerium oleander* L. occur along the drier lower reaches.

Agriculture (primarily extensive citrus groves) and grazing of goats and sheep have replaced the natural Mediterranean cork-oak (*Quercus suber* L.) woodland on the floodplain below the partially built dam (a cofferdam and an excavated subterranean flow diversion gallery). Observed impacts in the study area include diffuse organic pollution, nutrient enrichment, physical disturbance (riparian clearance, bank reinforcement) and reduced longitudinal connectivity (low-step damming and abstraction for irrigation by pumping from the riverbed). Urbanisation is relatively scant and *Eucalyptus globulus* Labill. and *Pinus pinaster* Aiton plantations are present in the higher areas of the basin. Tributaries suffer little physical disturbance, however the Monchique River receives organic input from piggeries and the small village of Monchique, whilst the lower reaches of the Monchicão tributary are affected by abstraction for agriculture. The Ribeira de Carvalho tributary is far less impacted.

Figure 3.1 Map of Odelouca basin with sampling points.

Field sampling and data collection

Habitat Assessment

Since Mediterranean systems are subject to flooding and drying (Gasith & Resh, 1999), where a cycle of abiotic (scouring and drying) and biotic/physicochemical factors (increased water temperature, increased levels of predation in dry summer pools) exert a notable influence on river communities, all fieldwork was carried out between spring and early summer (February – May 2006), when full flow connectivity existed and levels of habitat diversity and available resources were highest. Data were collected from a total of 30 sites (Fig 3.1): 25 sites along the main channel and five sites along the tributaries of Ribeira de Carvalho (1), Ribeira de Monchique (2) and the Ribeira de Monchicão (2).

Habitat structure, diversity and quality were assessed over a 500 m reach using an adapted version of the UK River Habitat Survey (RHS; addition of land use categories and plant species found on the Iberian Peninsula), a method that records substrata and flow type, natural features and modifications of the margins and river bed, land use, presence and complexity of riparian vegetation, together with measurements of stream and bank dimension (Raven *et al.*, 1997). Features were recorded at 10 spot checks situated at 50 m intervals and then assessed over the whole 500 m stretch ("sweep up"). Recorded features were used to assess the extent of natural or artificial features and characterise river habitat condition. Other important data on geology, climate, temperature, altitude, relief, land use, land cover, organic and industrial discharge and the presence of roads were obtained using GIS sources (the Portuguese National Water Institute - INAG, CORINE), aerial photography and downloadable catchment data from the INAG website (see Fernandes *et al.*, 2007).

Lotic communities

All organism groups considered in this study were sampled within each 500 m RHS reach, using European STAR project methodologies (except for the avifauna), modified for application in Portuguese lotic systems (manuals published by INAG downloadable at http://dqa.inag.pt/dqa2002/port/docs_apoio/nacionais.html).

Benthic macroinvertebrate samples were taken along a 50-100 m long stretch at the beginning of each 500 m reach using an adapted AQEM multihabitat sampling protocol (AQEM, 1999; Hering *et al.*, 2004; INAG, 2008a). The sampling area was selected in order to cover the greatest possible diversity of habitats representative of the reach, including (if present) a riffle and areas of deposition. Types and extent of habitats were visually estimated and six 1 m long sampling units of the most representative habitats were taken (0.25m x 0.25m handnet; habitats with < 5% cover were excluded). The composite sample was placed in a labelled plastic flask and fixed *in situ* using 4% formaldehyde. In the laboratory, samples were washed, sieved, sorted and identified using a low-power stereo microscope. All individuals were picked from the samples; subsampling was used when more than 200 individuals of a given taxon were present in the sample. Macroinvertebrates were identified to the lowest possible taxonomic level, which was mostly genus or species. Where higher levels of taxonomic resolution were not possible (*e.g*. for many Diptera) the AQEM protocol for taxonomic adjustment was applied (Hering *et al.*, 2002).

Fish were sampled using an adapted STAR methodology (INAG, 2008b) with an Electracatch International, SAREL model WFC7-HV electrofisher (applying 300V and maintaining a 3-A output to a 40 cm stainless steel anode). Sampling was carried out (area sampled was 20 times the mean width of the survey reach, minimum length 100 m starting at RHS spot check 1) by walking upstream in a zigzag pattern (Lyons, 1996) or sampling from a boat at sites over 1 m deep (Godinho, Ferreira & Santos, 2000). Captured fish, held in large plastic containers, were identified to species, counted, and immediately returned to the river.

Bird surveys were carried out during the nesting period, ensuring the maximum number of nesting species, at three equidistant points (250 m distance between points) including the riparian gallery and surrounding area (Bibby *et al.*, 2000). All birds observed or heard were recorded over a 10 minute period at each point. The distance of the bird from the observer was estimated when the distance was inferior to riparian gallery width. Birds were also surveyed on the flood plain approximately 100 m perpendicular to each point in the riparian gallery. All recorded birds were identified to species.

Macrophyte inventories were carried out in spring and early summer when the greatest number of species was present and water transparency and depth were most favourable for survey work (CEN, 2003) along 100 m reaches (at the start of the 500 m RHS reach), taking into account the percentage cover of each species in the sample area, evaluated as if observed macrophyte populations occurred as a single agglomeration (INAG, 2008c). Epiphytic bryophytes above the splash/humid zone were not included in the surveys. Species level determinations were made on site; specimens that could not be identified were taken to the Superior Agronomy Institute herbarium (Technical University of Lisbon) and bryophytes to the herbarium of the Botanical Garden (Science Faculty of Lisbon) for identification. A list of recorded taxa and metrics calculated for each group is given in Appendix 1.

At each macroinvertebrate and fish sampling site, physicochemical measurements of temperature, conductivity, pH and dissolved oxygen were taken with hand held electronic field probes. Depth was measured with a metre rule to the nearest centimetre and water velocity estimated with an ultrasonic flow meter (FP101 Global Flow Probe). Mean water velocity was measured at 0.6 total depth when total depth was < 0.8 m; otherwise velocity was measured at 0.2 and 0.8 of total depth (Bovee & Milhous, 1978).

Based on the literature and available data, a list of biological metrics was drawn up (Appendix 1) covering, enumeration, diversity measures, tolerance/intolerance measures, habitat preference traits (*e.g*. zonation, habitat or flow preferences), feeding traits, mode of existence (*e.g*. locomotion) and taxonomic groups (see Feld & Hering 2007). Macroinvertebrate metrics were calculated using the ASTERISC software (version 3, downloaded from the AQEM website http://www.aqem.de).

Data analyses

Redundant environmental and pressure parameters covering three spatial hierarchical levels (basin, reach and habitat) were removed using the Spearman Rank Correlation analysis method (Feld & Hering, 2007; Hering *et al.*, 2006a; Hering *et al.*, 2006c). If two environmental or pressure variables were highly correlated (threshold value of $r \geq 0.7$) the variable with a higher overall mean correlation coefficient was excluded from further analysis. A similar approach was used for the metrics calculated for each biological group (threshold of $r > 0.8$ or $r < -0.8$). Considerable numbers of highly correlated variables and metrics still remained after this process, so it was repeated with a threshold of $r \geq 0.5$ or $r \leq -0.5$ for environmental and pressure variables and $r \geq 0.6$ or $r \leq -0.6$ for metrics.

Table 3.1 Environmental (E) and Pressure (P) variables divided over three spatial scales (basin, reach and habitat) retained for PCA and RDA after analysis for redundancy following comparison of the Spearman Correlation Coefficient.

Variable & spatial scale		Unit/expression	Abbreviation	Source of data
Basin				
E	Valley form	Class 0–4	VAL_FRM	GIS
P	Urban area	% Catchment area	URB_A	GIS
P	Monocultures	% Catchment area	MONO_A	GIS
P	Agriculture	% Catchment area	AGRI_A	GIS
Reach				
E	Altitude	m.a.s.l	ALT	GIS
E	Number of bars	Count	BARS	RHS
E	Bankfull width (m)	Metres	BKFULL	RHS
E	Average riparian width	Metres	WDTRIP	RHS
E	Land use natural/semi-natural	Class 0–4 (% area)	LU250_NAT	RHS/GIS 250 m bankside buffer
P	Bank reinforced	Count derived score 0–4	BK_RI	RHS
P	Bank embanked	Count derived score 0–4	BK_EM	RHS
P	Land use agriculture	Class 0–4 (% area)	LU250_AG	RHS/GIS 250 m bankside buffer
Habitat				
E	pH	Sorensen scale	PH	Sample site
E	Dissolved oxygen	mg L^{-1}	DO	Sample site
E	Water velocity	m s^{-1}	WVEL	Sample site
P	Bank modification		BNK_MOD	RHS/sample site
P	Channel modification		CH_MOD	RHS/sample site
P	Banktop landuse agriculture	Class	BT_AG10	RHS/sample site 10 m bnkside buffer
P	Banktop landuse pasture	Class	BT_RP10	RHS/sample site 10 m bnkside buffer

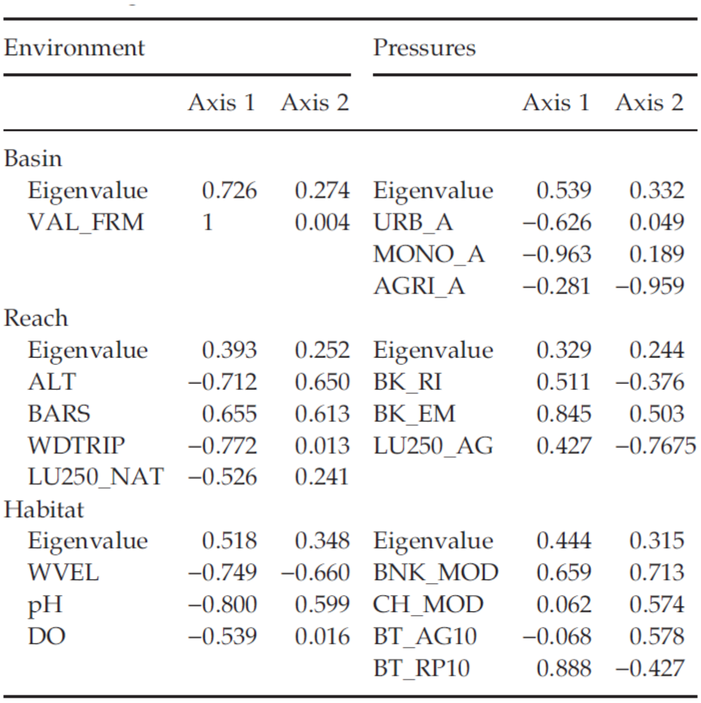
Principal Components Analysis (PCA) was carried out on environmental and pressure data sets at each spatial level to reduce data dimensionality and identify the principal environmental and pressure gradients (May & Brown, 2002). From an initial list of 38 environmental variables and 36 pressure variables, a total of nine environmental and 10 pressure variables were retained (Table 3.1). The same procedure was carried out for functional data (metrics) for each organism group. A total of 22 metrics were retained from an initial list of 125 metrics (Appendix III a) and a list of taxa for each organism group is given in Appendix IIIb.

Structural and functional data for each organism group (log (x+1) or arcsin square root transformed according to data type; singly occurring specimens and those occurring at less than three sampling sites excluded) were analysed using Detrended Correspondence Analysis (DCA) to assess the biological turnover or gradient within each data set and determine *a posteri* use of unimodal or linear constrained response analyses (Gauch, 1982a) Gradient lengths obtained for all organism groups (< 3.0 SD units) indicated the use of a linear model. Manual forward selection using a cut off point of $P > 0.1$ (Aguiar & Ferreira, 2005; Griffith *et al.* 2003; Magnan & Rodriguez, 1994) was used to retain non-redundant subsets of environmental and pressure variables explaining functional and structural data distribution patterns for each organism group. RDA was carried out on selected environmental and pressure variables for the functional and structural data of each organism group. A Monte-Carlo permutation test (999 permutations) on the first axis eigenvalue and the sum of all canonical eigenvalues or “trace” evaluated the significance of the environmental and pressure effects for each analysis. To further aid interpretation of results, Spearman rank correlations ($P < 0.05$) were calculated between retained organism group metrics/taxa (explained variance values ≥ 15%) and the samples scores of RDA ordination axes 1 and 2 (where most of the variance tends to reside). CANOCO 4.5 (ter Braak & Smilauer 1998) was used for DCA and RDA analyses and Statistica (version 6.0) for the calculation of correlation coefficients. For clarity, only taxa and metrics with explained variance values ≥ 15% are shown in the ordination plots (Feld & Hering, 2007).

3.4 Results

Environmental and Pressure Gradients

PCA indicated higher levels of explained variance at the highest and lowest spatial scales, followed by reach scale for both sets of variables (see Table 3.1 for details of retained variables and abbreviations and Table 3.2 for PCA results)



Environment			Pressures		
	Axis 1	Axis 2		Axis 1	Axis 2
Basin					
Eigenvalue	0.726	0.274	Eigenvalue	0.539	0.332
VAL_FRM	1	0.004	URB_A	−0.626	0.049
			MONO_A	−0.963	0.189
			AGRI_A	−0.281	−0.959
Reach					
Eigenvalue	0.393	0.252	Eigenvalue	0.329	0.244
ALT	−0.712	0.650	BK_RI	0.511	−0.376
BARS	0.655	0.613	BK_EM	0.845	0.503
WDTRIP	−0.772	0.013	LU250_AG	0.427	−0.7675
LU250_NAT	−0.526	0.241			
Habitat					
Eigenvalue	0.518	0.348	Eigenvalue	0.444	0.315
WVEL	−0.749	−0.660	BNK_MOD	0.659	0.713
pH	−0.800	0.599	CH_MOD	0.062	0.574
DO	−0.539	0.016	BT_AG10	−0.068	0.578
			BT_RP10	0.888	−0.427

Table 3.2 PCA eigenvalues and loadings (axes 1 and 2) for retained environmental and pressure variables divided over three spatial scales of habitat, reach and basin. Only variables with loadings ≥0.5 on least one of the first two PC axes are listed

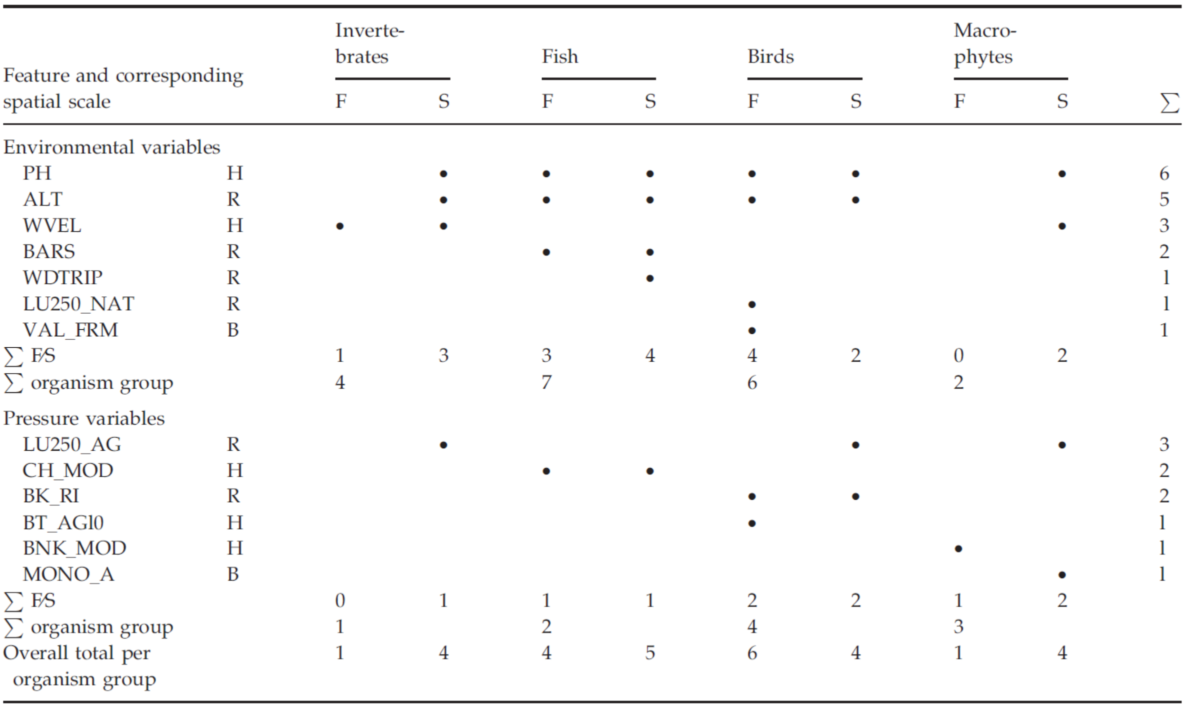
For clarity, only variables with loadings ≥ 0.5 are described. The first two axes of the basin level environmental PCA explained 100% of the variance (axis 1 =72.6%, axis 2= 27.4%), separating tributary sites and headwater sites situated in deep vee valleys from more heterogeneous mid section sites and shallow vee valley sites downstream of the cofferdam. The first two axes of reach level environmental variables (n=4; loadings ≥ 0.5) explained 64.5 % of the variance (axis 1 = 39.3%, axis 2 = 25.2%), separating sites with lower levels of human intervention situated higher up the river corridor from those below the cofferdam. Higher lying sites were characterised by riparian galleries, natural adjacent land use and instream habitat heterogeneity (the presence of bars). Habitat level PC axes explained 86.6% of the variance (axis 1 = 52.8%, axis 2 = 34.8%); the highest loadings along axes 1 and 2 were water velocity (WVEL_B, -0.749 and -0.660, respectively) and pH (PH, -0.800 and 0.599, respectively). Habitats in the middle and upper reaches were characterised by higher water velocity levels (WVEL_B), while lower lying sites had higher pH, temperature and dissolved oxygen levels (DO_F). Higher dissolved oxygen and to some extent even pH levels can be attributed to the considerable daytime photosynthetic activity of dense stands of macrophytes and filamentous algae recorded at lower sites.

Basin level PC analysis of pressures (total explained variance 87.1%; axis 1 53.9%, axis 2 = 33.2%) illustrated the effect of large scale changes in land use, attributed primarily to agriculture (MONO_A and AGRI_A). The first two axes of the reach level pressure PCA explained 57.3% of the variance. Loadings (≥ 0.5) along axis 1 clearly described physical bankside disturbance (BK_RI and BK_EM) and changes in adjacent land use (LU250_AG), mostly at sites on the floodplain below the cofferdam. Habitat level PCA (axes 1 and 2 explained 75.9% of the variance) indicated variables (loadings ≥ 0.5) describing physical disturbance including bankface/top and the river channel modification and agricultural and pastoral activities (grazing at the banktop).

Forward selection of variables

Forward selection created non-redundant subsets of variables describing higher levels of variance in relation organism group structure and function (Table 3.3). The number of variables shared between organism groups ranged from two to six; five selected variables were unique to the structural or functional data of a particular organism group.

Table 3.3 Results of the manual forward selection procedure of environmental and pressure variables relative to the functional (F) and structural (S) data for each organism group (spatial scale:B=basin, R=reach, H=habitat)



Feature and corresponding spatial scale		Invertebrates		Fish		Birds		Macrophytes		
		F	S	F	S	F	S	F	S	∑
Environmental variables										
PH	H		•	•	•	•	•		•	6
ALT	R		•	•	•	•	•			5
WVEL	H	•	•						•	3
BARS	R			•	•					2
WDTRIP	R				•					1
LU250_NAT	R					•				1
VAL_FRM	B					•				1
∑ F/S		1	3	3	4	4	2	0	2	
∑ organism group		4		7		6		2		
Pressure variables										
LU250_AG	R		•				•		•	3
CH_MOD	H			•	•					2
BK_RI	R					•	•			2
BT_AGI0	H					•				1
BNK_MOD	H							•		1
MONO_A	B								•	1
∑ F/S		0	1	1	1	2	2	1	2	
∑ organism group		1		2		4		3		
Overall total per organism group		1	4	4	5	6	4	1	4	

The variables pH (PH) and altitude (ALT) were highly redundant between all four organism groups. Forward selection retained more environmental variables than pressure variables for all organism groups, except macrophytes (environmental variables n=2, pressure variables n=3). A maximum of seven environmental variables were retained for fish, followed by six for birds, four for macroinvertebrates and only two (structural data) for macrophytes. More environmental variables were retained for organism group structural data, with the exception of avian data. Intra group redundancy was highest within the fish for pH (PH), altitude (ALT) and the presence of bars (BARS) and inter group redundancy was highest between fish and bird structural and functional data (pH and altitude). Fewer pressure variables were retained across the organism groups and intra group and inter group redundancy levels were lower. However, LU250_AG, indicating agricultural land use was selected for macroinvertebrate, avian and macrophyte structural data. Intra group redundancy was evident for channel and bankside disturbance (CH_MOD and BK_RI) for fish and bird groups. The highest number of retained pressure variables (n=4) was for birds followed by macrophytes (n=3). A single variable was retained for benthic macroinvertebrate structural data and none for functional data indicating that the selected metrics do not appear to reflect the principal detected pressures.

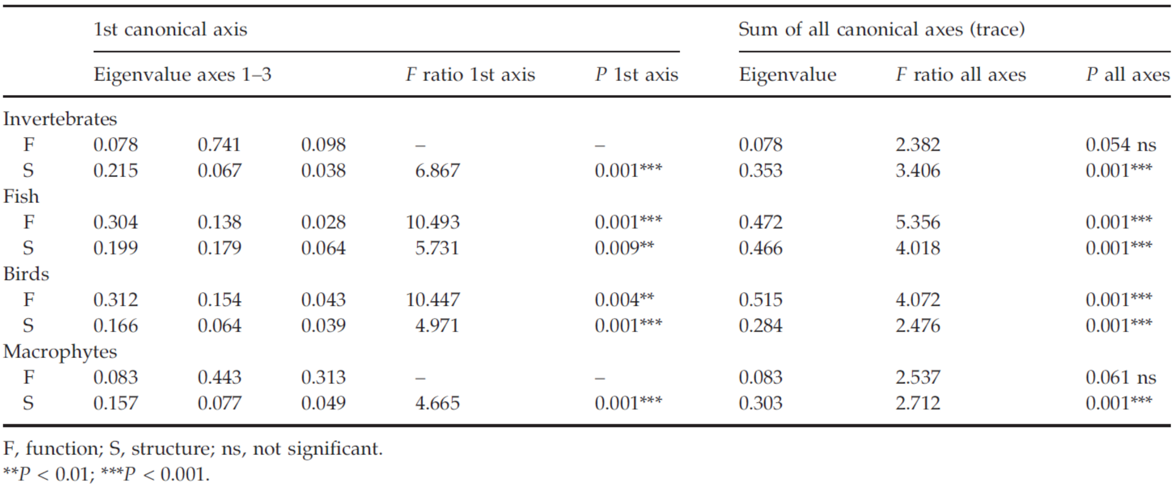
Most retained variables derived from reach scale (46.15%), followed by habitat (38.46%) and basin level (15.37%). Similar to Feld & Hering (2007), the larger number of selected reach level variables may highlight the importance of processes at this spatial scale in determining group structure and function or the pervasive effect of predominantly reach level RHS recorded variables in the dataset.

RDA results

The extracted first axis and trace for RDA ordinations were mostly highly significant (Monte Carlo test; Table 3.4), with the exception of benthic macroinvertebrate and macrophyte functional data. Fish and bird structural and functional data provided the highest axis 1 eigenvalues. RDA ordination plots revealed the relation between organism groups and retained variables (Fig. 3.2, Fig. 3.3).

Water velocity (WVEL) was the only variable retained for the macroinvertebrate functional RDA (Fig.2a). Metrics (> 15% explained variance) were negatively correlated with axis 1, indicating the presence of organisms with rheophilic current preferences (%_RHEOPHIL, r=-0.812), lithal microhabitat preferences (% LITHAL, r=-0.661), higher numbers of Trichoptera taxa (N_TRICH, r= -0.866) and EPT (Ephemeroptera, Plecoptera and Trichoptera) taxa (EPT, r= -0.821). Despite initial sorting using the Spearman Rank Correlation (see Feld & Hering, 2007; Hering *et al.*, 2006a; Hering *et al.*, 2006c), the proximity of the metrics in the ordination space suggests collinearity.

Table 3.4 Results of RDA and Monte Carlo permutation (F statistic, 999 permutation) for testing the significance of environmental and pressure variables retained by forward selection in relation to structural and functional data of benthic macroinvertebrates, fish, birds and macrophytes



	1st canonical axis					Sum of all canonical axes (trace)		
	Eigenvalue axes 1–3			*F* ratio 1st axis	*P* 1st axis	Eigenvalue	*F* ratio all axes	*P* all axes
Invertebrates								
F	0.078	0.741	0.098	–	–	0.078	2.382	0.054 ns
S	0.215	0.067	0.038	6.867	0.001***	0.353	3.406	0.001***
Fish								
F	0.304	0.138	0.028	10.493	0.001***	0.472	5.356	0.001***
S	0.199	0.179	0.064	5.731	0.009**	0.466	4.018	0.001***
Birds								
F	0.312	0.154	0.043	10.447	0.004**	0.515	4.072	0.001***
S	0.166	0.064	0.039	4.971	0.001***	0.284	2.476	0.001***
Macrophytes								
F	0.083	0.443	0.313	–	–	0.083	2.537	0.061 ns
S	0.157	0.077	0.049	4.665	0.001***	0.303	2.712	0.001***

F, function; S, structure; ns, not significant.
$^{**}P < 0.01$; $^{***}P < 0.001$.

The taxa based RDA (Fig.3.3a) indicated strong environmental/pressure gradients along axes 1 and 2, illustrated by the length of the arrows attributed to the ordinated variables. Axis 1 revealed a longitudinal pH (intra-set correlation r= -0.889) and altitude gradient (intra-set correlation r= 0 0.586), essentially representing the river continuum. Axis two separated faster flowing sites (WVEL, intra-set correlation r= -0.454) from slow flowing sites in areas affected by agricultural land use (LU250_AGR intra-set correlation -0.869). A longitudinal taxonomic distribution pattern showed that taxa occurring more abundantly in the lower reaches of the Odelouca (higher pH axis 1) included *Baëtis* sp. (r= -0.831), Chironomidae (r= -0.831), *Dugesia* sp. (r= -0.764), *Gyraulus* sp. (r= -0.0.688) and *Simulium* sp. (r= -0.849). Taxa at the other end of this gradient were *Capnioneura mitis* Despax (r= 0.469), Tanytarsini, and *Chironomus plumosus*-gr (both not significant). *Stylaria lacustris* L. (r= -0.822) and *Procloeon* sp. (r= -0.577) were strongly associated with the agricultural land use (axis 2), while *Dicronata* sp. (r= 0.680), *Atherix* sp. (r= 0.670) and *Onchychogomphus forcipatus* L. (r= 0.653) were associated with higher water velocity.

Axis 1, revealed as a longitudinal pH gradient (intra-set correlation r= -0.765), was also evident for fish functional data (Fig. 3.2b), while axis 2 represented changes in instream properties from less disturbed higher reaches (ALT intra-set correlation -0.650, BARS intra-set correlation -0.394) to lower lying reaches affected by channel modification (CH_MOD intraset correlation 0.546). Fish metrics clearly separated sites along this gradient, describing shifts in exotic or native species status and feeding or habitat preferences. No diversity indices were selected, a facet of the harsh Mediterranean intermittent

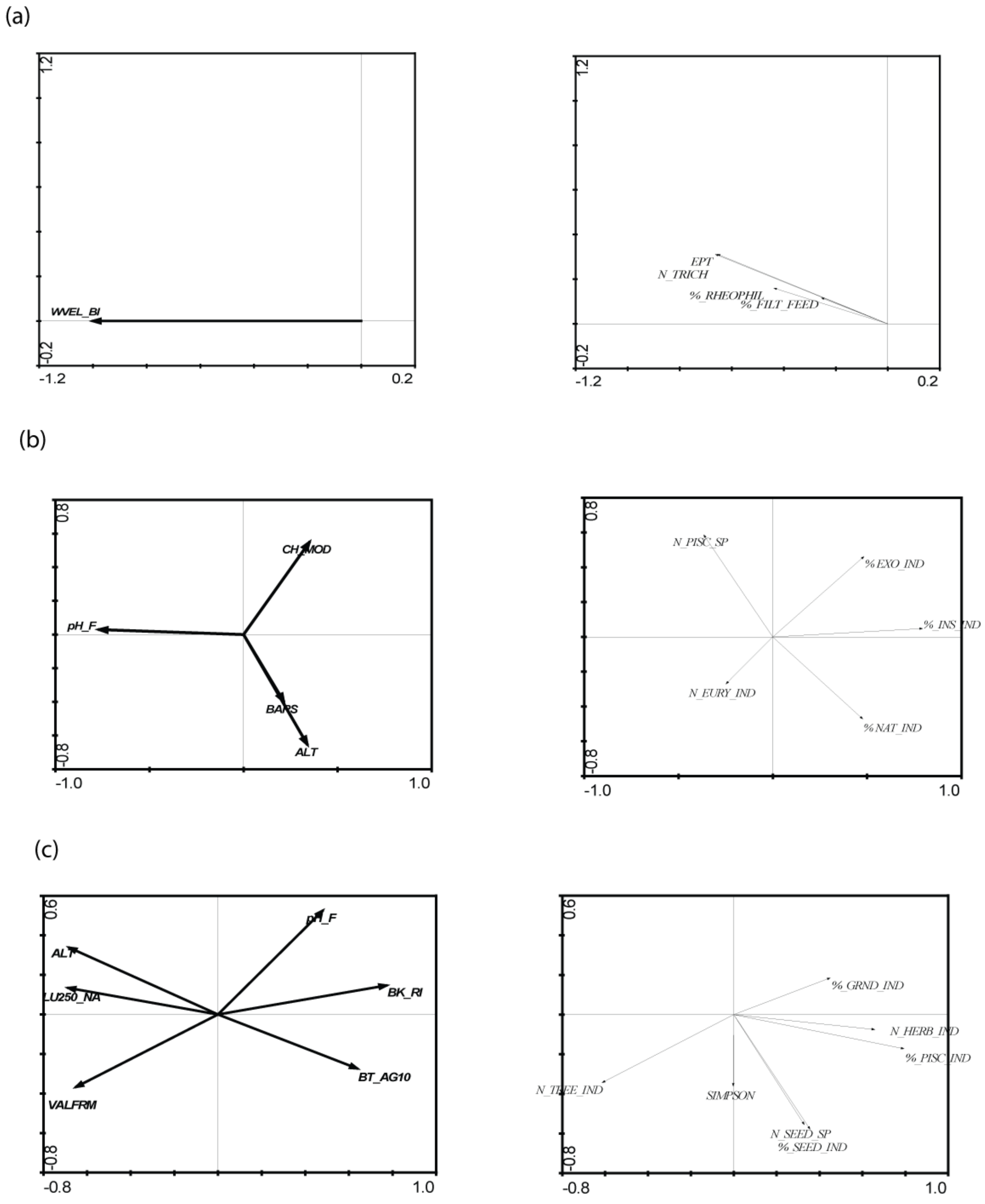


Figure 3.2. RDA ordination plots of environmental and pressure variables and (a) macroinvertebrate functional data (b) fish functional data (c) bird functional data. Only functional organism group descriptors with an explained variance > 15% are shown (no macrophyte functional descriptors have explained variance values >15%).

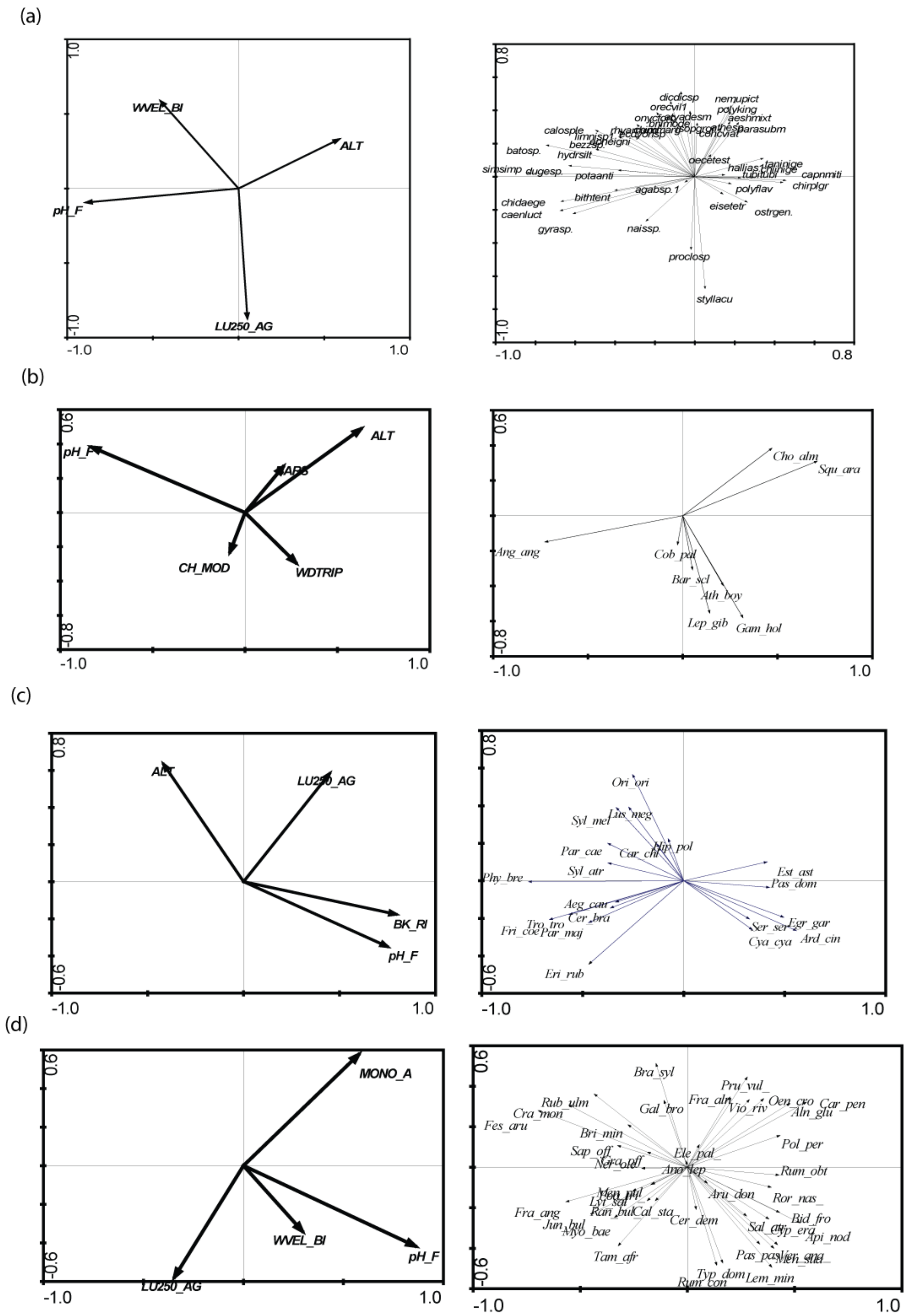


Figure 3.3 RDA ordination plots of environmental and pressure variables and (a) macroinvertebrate structural data (b) fish structural data (c) bird structural data. Only taxa with an explained variance > 15% are shown.

habitat templet resulting in depauperate fish assemblages (Magalhães *et al.,* 2002). The percentage of insectivore fish species (r=0.969) clearly separated sites along axis 1, while metrics describing percent exotic and native individuals reflected physical habitat change (r=0.666 and r=-0.868, respectively). Piscivorous species were negatively associated with less disturbed sites.

Patterns in fish structural data closely mirrored metric distribution patterns, since metrics were derived from a small number of ecologically distinct species (Fig. 3.3b). Axis 1 also described a longitudinal pH gradient, while axis 2 described a shorter gradient related to channel modification and riparian gallery width (intraset correlation CH MOD= -0.228, WDTRIP = -0.292). Threatened native species were clearly associated with instream habitat quality and negatively associated with downstream habitat degradation (axis 1: *S. aradensis,* r=0.763, *C. Almacai,* r= 0.51*;* axis 2: *S. aradensis,* r=0.533, *C. almacai,* r= 0.413). Although widespread, the eel *Anguilla anguilla* L. was more abundant at lower lying sites (axis 1, r=-0.770). A mixture of exotic (*Gambusia holbrooki* Girard and *Lepomis gibossus* L.) and native (*Atherina boyeri* Risso, *Barbus sclateri* Günther and *Cobitis paludica* de Buen) species were associated with channel modification and riparian width. The presence of the native Andalusian barbel, *B. sclateri* reflected its preference for pool-like conditions (Magalhães *et al.,* 2002) within the vicinity of the cofferdam, while the exotic *G. holbrooki* comprised just over 50% of the total catch at sites close to the cofferdam, where flow conditions were much reduced but riparian galleries were still present.

Bird functional data (Fig. 3.2c) strongly separated sites along distinct environmental and pressure gradients (axis 1) related to land use and bankside physical impacts. Most retained metrics described either feeding or habitat preferences; only a single diversity index (Simpson) was retained. Parameters on the left side of the biplot (Fig. 3.2c) separated higher lying sites with natural land use situated in asymmetrical or narrower valleys (intraset correlatons ALT = -0.681, LU250NAT = -0.685, VALFRM = -0.649) with good riparian stands, reflected in the metric describing the number of tree dwelling individuals (N_TREE_IND). Parameters on right side of the biplot separated lower lying sites subject to physical impacts such as bank reinforcement (intraset correlation BK_RI = 0.771 and BT_AG10 = 0.636). Associated metrics included the number of herbivore individuals (N_HERB_IND, r=0.564) and the percentage of piscivorous individuals (%_PISC_IND, r= 0.672). Metrics describing seed eating species and individuals along axis 2 tended to be higher in areas affected by agriculture (N_SEED_SP, r= -0.789; %_SEED_IND, r=-0.659), reflecting a change in vegetation resulting from riparian clearance and the natural presence of more open ground. The number of tree dwelling individuals was highly correlated with both axes 1 and 2 (N_TREE_IND, axis 1, r= -0.692, axis 2, r= -0.585).

Avian structural data revealed distinct groups of species (Fig. 3.3c). The longitudinal gradient from higher lying sites to lower lying degraded sites was discernible (axis 1 intraset correlation: pH, r= 0.750, BK_RI, r= 0.794; axis 2 intraset correlation: ALT, r=0.633, LU250_AG, r=0.582). Species clearly associated with bankside disturbance (reinforcement) at lower lying sites were the Grey Heron *Ardea cinerea* L. (axis 1, r=0.452), the little Egret *Egretta garzetta* L. (axis 1, r=0.557) and the house sparrow *Passer domesticus* L. (axis 1, r=0.557). Woodland species negatively associated with agricultural areas were the Iberian Chiffchaff *Phylloscopus brehmii* von Homeyer, the chaffinch *Fringilla coelebs* L. (axis 1, r=-0.706), the Great Tit *Parus major* L. (axis 1, r=-0.516) and the Northern Wren *Troglodytes troglodytes* L. (axis 1, r=-0.825). A third group of predominantly woodland/scrub species was also associated with less impacted sites situated further upstream. Species significantly correlated with axis 2 were the Nightingale *Luscinia megarhynchos* Brehm (axis 3, r=0.641), the golden Oriole *Oriolus oriolus* L. (axis 2, r= 0.781) and the Sardinian Warbler *Sylvia melanocephala* Gmelin (1axis, r=0.518).

No macrophyte metric explained more than 15% variance relative to the single retained pressure variable describing bank modification, hence no figure is given. Macrophyte structural data revealed distinct distribution patterns related to longitudinal changes in the hydric regime and habitat degradation associated with land use (Fig. 3.3d). The longitudinal pH gradient (pH, axis 1 intraset correlation, r= 0.859; axis 2, intraset correlation, r= -0.416) and land use gradient was evident (LU250_AG, axis 1 intraset correlation, r= 0.344; axis 2, intraset correlation, r= -0.576; MONO_A, axis 1 intraset correlation, r= 0.576; axis 2 intraset correlation, r= 0.576). Woody riparian species tolerant of wetter conditions, such as the common alder *Alnus glutinosa* (L.) Gaertner (axis 1, r=0.577; axis 2, r=0.484), the glossy Buckthorn *Fraxinus alnus* P. Mill (axis 2, r=0.495), the dog violet *Viola riviniana* (Rchb.) and Common Selfheal *Prunella vulgaris* L. subspecies *vulgaris* (axis 2, r=0.450), were associated with tributary sites where narrow strips of monocultural cereal crops (MONO_A) were present beyond the riparian zone. Woody shrubs tolerant of arid and disturbed conditions such as *Tamarix africana* Poiret (axis 1, r=0.368; axis 2, r=0.663) the Desert or Narrow Leaved Ash *Fraxinus angustifolia* Vahl. (axis 1, r=-0.804) and *Juncus bulbosus* L. (axis 1, r= -0.514) were present at sites affected by agriculture. Degraded lower lying sites were characterised by the instream presence of false water cress *Apium nodiflorum* (L.) (axis

1, r= 0.671) and Common Duckweed *Lemna minor* L. (axis 2, r=-0.505) and common tickseed *Bidens frondosa* L. (axis 1, r= 0.766), *Cyperus eragrostis* Lam (axis 2, r=- 0405) on the bankside. Species at main channel sites with lower levels of human intervention included woody shrub species such as the bramble *Rubus ulmifolius* Scott (axis 1, r=-0.618), the common Hawthorn *Crataegus monogyna* Jacq (axis 1, r=-0.509) and *Festuca arundinacea* Schreber (axis 1, r=-0.749).

3.5 Discussion

The Habitat Templet Theory (Southwood, 1977; Southwood, 1988) describes organism structural and functional traits as a reflection of contemporary habitat conditions. Integrated multiple organism approaches use responses from a range of different life histories under certain habitat conditions to provide an early warning system to particular pressures with hopefully little or no redundancy (Furse *et al.*, 2006, Hering *et al.*, 2006b). Ideally, organism group response to stressors should be type specific and easily distinguishable from responses to environmental gradients (Johnson *et al.*, 2006a). The results of this study highlight important considerations in selecting suitable indicator organism groups for effective biomonitoring programmes, such as river typology, the identification of environmental gradients, the types of impact affecting the system under study and links between organism groups and stressors (Hering *et al.*, 2006a; Hering *et al.*, 2006c; Resh, 2008).

PCA revealed scale dependent environmental and pressure factors and potentially high levels of covariance (Allan, Erickson & Fay, 1997; Allan, 2004; Allan & Johnson, 1997; Hawkins *et al.*, 2000; Poff, 1997 Frissell *et al*., 1986). European Mediterranean river basins have a long history of human disturbance from changes in land use on the floodplain to management of the channel and riverbanks for flood protection and to increase water supply. The history of human activity in Iberian Mediterranean systems, combined with the natural temporal and spatial fluvial patterns in these systems (Gasith & Resh 1999) obscures the ready distinction of these two sources of variability (Díaz, Suárez Alonso & Vida-Abarca Gutiérrez, 2008; Hooke, 2006).

A purported advantage of the multi organism approach is that the range of responses distinguishes different types of stressors that are ideally distinct from responses to natural phenomena. Redundancy analysis clearly showed environmental factors to be major drivers of change in group structure and function, directly and indirectly describing essentially longitudinal distribution patterns, resulting in high levels of intra and intergroup redundancy for parameters such as pH and altitude. Ostensibly classified as an environmental factor in this study, the chemical pH gradient illustrates the combined effect of multiple mechanisms operating over several spatial scales as a result of underlying natural biotic and abiotic gradients, changes in land use and enrichment. Background pH levels are normally determined by catchment geology and biotic processes such as decomposition, photosynthesis and respiration (Townsend, Hildrew & Francis, 1989). Less shaded lower reaches of the Odelouca (increased bankfull width, riparian clearance and changes in land use) were characterised by dense aquatic macrophyte stands of false water cress *Apium nodiflorum* (L.), *Potamogeton* and filamentous algae (recorded using RHS). Higher pH levels were almost certainly partly attributable to increased runoff and photosynthetic activity which influenced instream physicochemical processes. A prior study on pH gradients and land use in Southern English streams has described the reason for the strong pH gradient and its obvious influence upon macroinvertebrate community structure as “undoubtedly complex” (Townsend *et al.*, 1989).

Retained pressure variables described changes in land use or physical disturbance of the bankside or channel. The number of pressure variables retained and levels of redundancy were lower and it became clear that benthic macroinvertebrate and macrophyte based metrics were not reliable indicators of the physical sources of disturbance recorded in the study area. By contrast, Feio *et al.* (2007) found macroinvertebrate assemblages in the Mondego River, a transitional river system in central Portugal, to be sensitive to morphological bankside and channel disturbance. These different responses are almost certainly related to typology. The Mondego is a predominantly perennial river system compared to the highly intermittent Mediterranean character of the Odelouca, where benthic macroinvertebrate communities are naturally tolerant of periods of physical disturbance such as scouring caused by winter rainfall runoff followed by drought. Hering *et al.* (2006b) consider benthic macroinvertebrate communities to be unsuitable for monitoring hydromorphological degradation in southern European rivers due to the lack of taxonomical knowledge. However, macroinvertebrate structural distribution patterns were clearly related to large scale changes in land use and water velocity, a vital environmental variable that will be drastically altered upon completion of the Odelouca dam.

Mediterranean flow patterns strongly affect macroinvertebrate communities; water velocity was shown to be an important environmental variable for benthic macroinvertebrate structure and function in this study. Marked shifts in benthic macroinvertebrate taxonomic and biological traits, community structure and abundance have been found in long term studies of wet and dry years in intermittent Mediterranean systems (Bêche, McElravy & Resh, 2006, Bêche & Resh, 2007b; Bonada *et al.*, 2007) and in Mediterranean systems with permanent, intermittent and ephemeral flow regimes (Bonada *et al.*, 2007). Bonada *et al.* (2007) also found a predominance of invertebrate taxa with pool like strategies in intermittent streams (such as the Odelouca), highlighting the functional constraints imposed by the Mediterranean habitat templet. Further, findings from the structural data could be used to develop new, possibly trait based, metrics specific to the study area.

Although macroinvertebrate metrics did not successfully detect the physical impacts acting upon the Odelouca, macroinvertebrate metrics in Rapid Bioassessment Protocols (RBP's) have successfully detected water quality impairment in Iberian and South African Mediterranean streams (Bonada *et al.,* 2006). Macrophyte functional data also did not successfully reflect community changes along the PCA pressure gradients. Forward selection retained a single pressure variable (bank modification) and metrics described only very small amounts of variance. Similar to findings from other Mediterranean systems such as the Tagus River (Ferreira & Moreira, 1999; Aguiar & Ferreira, 2005), macrophyte structural data clearly indicated longitudinal species distribution patterns related to changes in land use, water velocity and water availability. Flow and substratum heterogeneity resulting from flood events are fundamental to riparian seed dispersal, recruitment (Dixon 2003) and habitat heterogeneity which ultimately favour more trophically complex lotic communities (Pearson, Li & Lamberti,. 1992) Aquatic macrophyte and woody riparian communities play a vital role in increasing habitat diversity and creating aquatic refugia for invertebrates and juvenile fish (Pinto *et al.*, 2006) and riparian birds (Jansen & Robertson, 2001b). Important stands of riparian woody plants species were identified along the tributaries and main channel sites (which will be inundated once the dam is completed) where levels of human intervention were low. Woody riparian species in the lower reaches comprised species typically resistant both to drought and human disturbance, highlighting again the innate covariance that exists between natural factors and stressors in the river landscape. The permanent change in flow patterns caused by the construction of the Odelouca dam will profoundly affect macroinvertebrate and macrophyte community structure and function. This must be redressed via appropriate compensatory measures, such as the establishment of a typologically suitable annual environmental flow regime.

The results of this study agree with the findings of Hering *et al.* (2006) in linking larger, more mobile organisms that occupy a range of habitats with environmental and pressure factors operating at higher spatial scales. Results clearly emphasise the importance of birds as indicators of wider river corridor health together with fish as more direct indicators of river health. Marked structural and functional distribution patterns were linked to the major PCA gradients, namely changes in land use, physical impacts to the bankside and channel and habitat quality. Fish community structural and functional changes have been marked following the construction of the cofferdam and flow diversion tunnel between 2001 and 2003. Surveys prior to construction recorded *S. aradensis* and *I. almacai* as the two most abundant species, occurring along the entire length of the Odelouca (Pires *et al.,* 2004). RDA ordination biplots from this study clearly illustrate how these species are now confined to the upper reaches where habitat heterogeneity is greater, that the eel *A. anguilla* is more abundant in the lower reaches and that exotic species, previously described as having “no relevance for the fish assemblage”, are far more prevalent (Pires *et al.,* 2004) in areas suffering channel modification. Increased incidence of exotic species due to habitat degradation has also been observed in the Guadiana catchment, another Portuguese Mediterranean river system where eight native high conservation status species are known to occur (Bernardo *et al.,* 2003; Collares-Pereira *et al.,* 2000). Mediterranean fish communities tend to be depauperate but highly endemic because of the harsh Mediterranean regime. Native species are well adapted and respond via migration and/or tolerance of residual surface water refugia during the dry season to avoid the drastic but predictable changes in habitat conditions (Magalhães *et al.*, 2002; Bernardo *et al.*, 2003; Davey & Kelly, 2007; Magoulick & Kobza, 2003). Flexibility in habitat use as a response to the strong changes that characterise Mediterranean rivers is another tactic to increase overall survival by avoiding unfavourable habitats, *e.g*. those prone to scouring during flood or drying during the summer (Godinho, Ferreira & Cortes, 1997; Godinho *et al.*, 2000; Moráno-López *et al.*, 2006). In a study on the effect of multi-year droughts on fish assemblages in the Torgal stream, situated only 40 km from the Odelouca study area, Magalhães *et al.* (2007) found little change in species richness and composition metrics but significant variation in individual species abundances in relation to interannual cycles of summer drought severity and the occurrence of rainy springs. Our results from a single snap-shot study confirm the limited application of species richness or diversity metrics and highlight the importance of measures related to the abundance of native or exotic species and feeding or habitat preference. However, the latter two traits may vary seasonally and require further study.

Avian structural and functional distribution patterns were strongly associated with habitat quality, changes in landuse and physical disturbance to the bank profile, highlighting the important role the avian community plays in direct riparian quality assessment along the Odelouca. Bird assemblages have been used as indicators of woodland and riparian condition (Bryce *et al.* 2002; Hinsley *et al.*, 2008), large scale changes in land use (Ormerod *et al.*, 2000) and hydromorphological gradients (Vaughn *et al.*, 2007b). Further, studies on dippers in relation to acidification (Ormerod *et al.*, 1986) and water quality (Feck & Hall Jr, 2004) have clearly illustrated the link between riparian birds and instream degradation resulting from global and local processes. Although adopting a different analytical approach, Vaughn *et al.* (2007b) used RHS data and bird census data to identify river bird distribution patterns and longitudinal environmental gradients linked to flow type, bankside and channel vegetation, depositional features and manmade structures. Bryce *et al.* (2002) used bird based metrics to develop a riparian bird integrity index for an integrated assessment approach including fish and invertebrate indicators and watershed data. The strong response of bird functional data to bank side/riparian habitat fragmentation has also been observed in another Mediterranean river system, the River Sado (Moreira, Saraiva & Pinto, 1997) and in Australian river systems (Jansen & Robertson 2001b). Our results further emphasise the importance of bird communities as effective indicators of the ecological integrity of the wider river landscape (Vaughn *et al.*, 2007b).

This study across several communities was carried out over a single season (spring) in a single year during conditions of full connectivity. Annual and interannual cycle of flooding and subsequent drought in Mediterranean systems influence spatial and temporal cycles in fluvial, biotic and abiotic processes (Gasith & Resh, 1999; Hooke, 2006), with long term implications for biological communities. These predictable natural processes strongly influence organism group traits, a fundamental tenet of the River Habitat Templet (Townsend & Hildrew 1994; Magalhães *et al.* 2002; Bernardo *et al.* 2003; Bêche & Resh 2007a; Bêche & Resh 2007b; Bonada *et al.* 2007; Díaz, *et al.* 2008) and have implications concerning the accurate interpretation of community responses to perturbation. These factors must be taken into account for developing and implementing effective mitigation and compensation measures during and following dam construction.

This study has shown how selected organism group functional and structural data can be used to detect redundancy and establish typologically appropriate links with stressors and environmental factors in an intermittent Mediterranean river system. It has also shown that detected gradients can comprise a complex interaction between natural and manmade factors. In the case of the Odelouca, fish and bird metrics and species appear to be reliable indicators of the large scale physical impacts, such as riparian habitat fragmentation, physical impacts to the riverbank and channel and flow disruption. Macroinvertebrate and macrophyte species distribution patterns in relation to flow and large scale change in land use can be applied in the testing and development of a multi-organism, multimetric system (Hering *et al*., 2006a) to assess the ecological condition of the Odelouca and the effect of future mitigation and compensation measures. The permanently compromised flow regime in a system previously defined by dramatic, seasonal patterns of flood and drought must be taken into consideration once the dam comes into operation. This permanent impact will undoubtedly affect all the organism groups considered in this study, emphasising the importance of temporal studies for the development of appropriate mitigation and compensation measures.

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Chapter 4

THE USE OF RIPARIAN STRIPS AS ECOLOGICAL CORRIDORS BY SONGBIRDS: AN EXPERIMENTAL STUDY PRIOR AND AFTER CLEAR CUT ACTIONS

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4. The use of riparian strips as ecological corridors by songbirds: an experimental study prior and after clear-cut actions

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4.1 Abstract

We conducted a two-year study during fall migration before and after a river valley clear-cut prior to dam exploitation in a Mediterranean-type river in Southern Portugal. Our aim was to study bird use of the riparian forest as corridor and how this could be impacted by removal of riparian habitat. The study was designed to assess if birds selected riparian strips and matrix differently, how habitat use changed after clear-cut actions and if these habitat changes affected the physical condition of birds. We used mist nets placed perpendicular and parallel to the river to infer the use of riparian forest as corridor. Matrix habitat was also surveyed. To control differences between years we used data from a ringing station where no habitat changes occurred between years. Results indicate that richness and abundance of bird assemblages were significantly higher in the riparian strip than in the matrix in both years. Significant higher numbers and species of birds were captured moving lengthwise to the river and more migratory birds were captured in the riparian forest in both years. After clear-cut: richness and abundance significantly decreased in all habitats; the number of days migrants remained in the area was significantly lower and fat scores and mean weight were highly significant lower. To our knowledge these findings have never been reported and suggest that poor quality riparian habitats might act as ecological traps and this should be considered when designing mitigation measures, restoring riparian corridors and developing ecological networks. Our results support the idea that riparian habitats act as corridors for resident and migratory birds.

4.2 Introduction

Wilson & Willis (1975) have proposed the development of ecological corridor networks to mitigate the impacts of habitat fragmentation. Despite several scientists argued that experimental evidence of efficacy of ecological corridors was virtually inexistent (Simberloff & Cox, 1987; Simberloff *et al*., 1992; Meffe & Carrol, 1994, Mech & Hallet, 2001) and ambiguous (Nicholls & Margules, 1991; Saunders & Hobbs, 1991), empirical evidence of negative impacts is rare (Haddad *et al*., 2011). The number and rigor of studies on the positive effects of corridors is increasing and providing evidence that corridors are valuable conservation tools (review in Beier & Noss, 1998; Tewksbury *et al*., 2002; Haddad *et al*., 2011; LaPoint *et al*., 2013).

Riparian forests have long been considered ecological corridors for resident, breeding and migratory birds but few quantitative accounts in the literature support this idea and some even mention that corridors may not be important for vagile species like birds (Gilbert-Norton *et al*. 2010; Cushman *et al*., 2013). On the other hand, recent studies have demonstrated the importance of riparian areas as ecological corridors for specific bird species in the tropics (Gillies & St. Clair, 2008; Sekercioglu 2009) and for communities of American resident and migratory birds (Skagen *et al*., 1998; Means & Finch, 1999; Finch & Young, 2000; Skagen *et al*., 2005; Mosley *et al*., 2006).

The clear-cut of areas to be flooded by dam operation is a common management practice with effects on river biodiversity (Cortes *et al.,* 1998; Santucci *et al*., 2005; Blakely *et al*., 2006; Catalano *et al*., 2007), causing habitat destruction and fragmentation. It is also an opportunity to evaluate the use of riparian forests as ecological corridors, since we can study if immigration to the target patch via corridor will be greater than if the corridor were absent (following Rosenberg *et al*., 1995). Forman & Godron (1986) defined corridor as a narrow strip of land that differs from the matrix on either sides, while Hess & Fisher (2001) identified six corridor functions: habitat, conduit, filter, barrier, source and sink/trap.

Our primary objective was to study the potential use and functions of riparian forests by songbirds for the design of mitigation measures and the development of effective corridors. In other words, corridors that do not constitute ecological traps (Schalaepfer *et al*., 2002) or barriers to bird survival and that promote connectivity between fragmented habitat and populations. We studied the use of strips of riparian forests as linear landscape structures, compared it with the matrix and analysed how the removal of riparian forests affects bird assemblages, namely on resident, short distance and long distance migrants (exploring habitat, conduit and sink/trap functions). We predicted that i) riparian forest supported higher levels of species diversity and abundance than the matrix (habitat function), ii) migrant species were more often detected in the riparian forest than in the matrix, thus using them as ecological corridors (conduit function), iii) the use of the riparian forest as core habitat was greater than the use as interface habitat, iv) diversity and species abundance would decrease for both residents and migrants after clear-cut (habitat and conduit function), iv) bird condition would be lower after clear-cut (sink/trap function).

4.3 Methods:

Study area

We conducted our study in the mid-section of the Odelouca River basin (511.4 km^2), a sub-catchment of the Arade basin (987.37 km^2), situated in Algarve, south west Portugal (Fig. 4.1). It is a medium-sized, low-gradient, incised lowland stream. After dam building 21 km of the Odelouca River were submerged, mostly in the mid-section of the basin. The dam embankment has a reservoir of 7.8 km^2. Prior to dam construction the river had continuous and floristically diverse riparian forests. Surrounding habitats in the mid-section of the river were dominated by Mediterranean scrubland and cork oak (*Quercus suber*) forested areas.

Climate is Mediterranean type, with annual rainfall following a seasonal pattern (wet season from October to March, dry season from June to August), resulting in a relatively slow running river subject to “flashy” discharge peaks during the winter. In the summer it runs dry, leaving temporarily unconnected pools in the river bed.

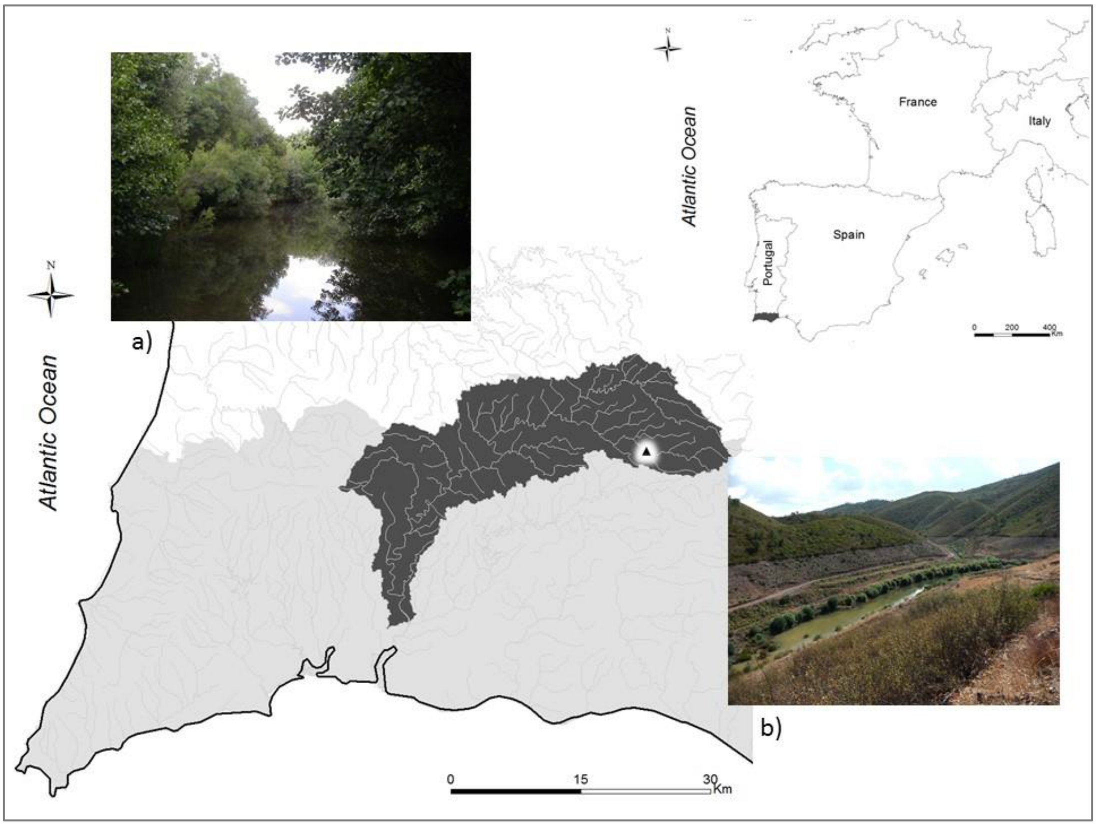

Figure 4.1. Location of the of the sampling area in Odelouca river basin, Portugal, Europe; a) detail of Odelouca river mature riparian forest; b) overview of Odelouca river riparian forest 9 months after clear cut; ▲location of the two mist netting stations

Clear-cut operations took place between March and December of 2009, with an interruption between May and June to reduce the impact during the breeding season. Vegetation clear-cut was set at 100 m from riverbed. In all, a total of 466 ha were clear-cut.

Habitat characteristics

We assessed habitat features (Table 4.1) along a 500 m reach using an adapted Iberian Peninsula version of the River Habitat Survey (Raven *et al*., 1997). We estimated vegetation cover visually for all vertical strata. Channels were steep (>45º) and cobble was the dominant channel substrate. No perceptible flow was registered and marginal dead water was present. We characterised channel vegetation by the presence of emergent reeds, sedges, rushes, grasses, horsetails and submerged plants. Before clear-cut the gallery was characterized by a continuous, dense and mature tree cover, dominated by alder (*Alnus glutinosa*), willow (*Salix salvifolia*), ash (*Fraxinus angustifolia*), oleander (*Nerium oleander*) and tamarisk (*Tamarix africana*). After clear-cut, new sprouts emerged in tree stumps in the following spring (2010), and some of these new sprouts reached 1 m height, forming isolated and scattered riparian spots. Major changes in channel after clear-cut include the lack of shading and a decrease in the water width.

The following habitat features were registered in upland habitats: land use within 50 m and 150 m, bush height, percentage of ground vegetation cover by vegetation (estimated visually), number of strata and dominant bush species. Upland habitats include the occurrence of natural Mediterranean bushland and cork-oak woodland. Before clear-cut, 90% of ground cover was colonized by rockrose (*Cistus ladanifer*) and strawberry tree (*Arbutus unedo*) with an average height of approximately 1 m, while after clear-cut sprouts cover 20%, showing roughly less than half initial height.

Table 4.1. Habitat characteristics in ringing sites located in River Odelouca before (2007) and after (2010) clear-cut operations.

Habitat type	Attributes	Site 1		Site 2	
		Before clear-cut	After clear-cut	Before clear-cut	After clear-cut
Channel	Banktop height (m)	1.7	1.7	3.5	3.5
	Bankfull width (m)	20	20	27	27
	Water width (m)	12	10	14	12
	Water depth (m)	0.4	0.4	0.3	0.3
	Number of riffles	7	5	2	2
	Number of pools	3	4	5	5
	Bank modification	resectioned (reprofiled)	resectioned (reprofiled)	natural	natural
	Channel features	exposed boulders	exposed boulders	exposed boulders and bedrock	exposed boulders and bedrock
	Shading of channel	present	absent	present	absent
Gallery	Width of riparian gallery (m)	3.5	1	3.5	1
	Extent of trees	continuous	Isolated/scattered	continuous	Isolated/scattered
	Tree height (m)	14	1	12	1
	Cover (%)	85	20	80	20
	Number of strata	3 (trees, shrubs and grass)	2 (shrubs and grass)	3 (trees, shrubs and grass)	2 (shrubs and grass)
	Features of special interest	Leafy debris and fringing reed-banks	Absent leafy debris but presence of fringing reed-banks	Leafy debris and fringing reed-banks	Absent leafy debris but presence of fringing reed-banks
Upland	Land-use within 50 m of banktop	Scrub & shrubs	Scrub & shrubs	Scrub & shrubs	Scrub & shrubs
	Land-use within 150 m of banktop	Scrub & shrubs & montado	Shrubs and rough unimproved grassland	Scrub & shrubs & montado	Shrubs and rough unimproved grassland
	Shrub height in upland (m)	1	0.5	1	0.5
	Cover (%)	90	20	90	20
	Number of strata	2 (trees and shrubs)	2 (shrubs and grass)	2 (trees and shrubs)	2 (shrubs and grass)
	Dominant shrub species	*Cistus ladanifer, Arbustus unedo*	*Cistus ladanifer, Arbustus unedo*	*Cistus ladanifer, Arbustus unedo*	*Cistus ladanifer, Arbustus unedo*

Bird data and mist-netting

We conducted the study during the fall migration of 2007 and 2010, before and after Odelouca river valley clear-cut. Two mist-netting sites with similar habitat features in the channel, margins and upland were selected, 1 km apart from each other to ensure that most individuals were not caught at more than one location (*e.g*. Ralph *et al*., 2004). Two different habitat types were sampled in each site: the riparian forest and upland habitat.

The riparian habitats were sampled in two different orientations: (1) perpendicular to the river, i.e. in the channel (to detect lengthwise movements) and (2) parallel to the watercourse (lateral dimension) (Fig. 4.2 – adapted version of Mosley *et al*., 2006). Our assumption is that birds caught in nets placed perpendicular were flying along the river, eventually using it as a corridor for movement (conduit function); birds caught in parallel mist-nets were flying from the channel to riversides or reverse thus using it as interface habitat broad sense (Naiman & Décamps 1997).

Mist netting in the two sites was carried out in both years during 13 days from 14th of September to 2nd of October. The net scheme used is summarized in Table 4.2. In order to catch birds moving through the understory we used net lanes as wide as the channel and the riparian gallery. Following Ralph *et al*. (2004) recommendations net lanes at upland were lower to cope for net exposure (1 and 2 shelves).

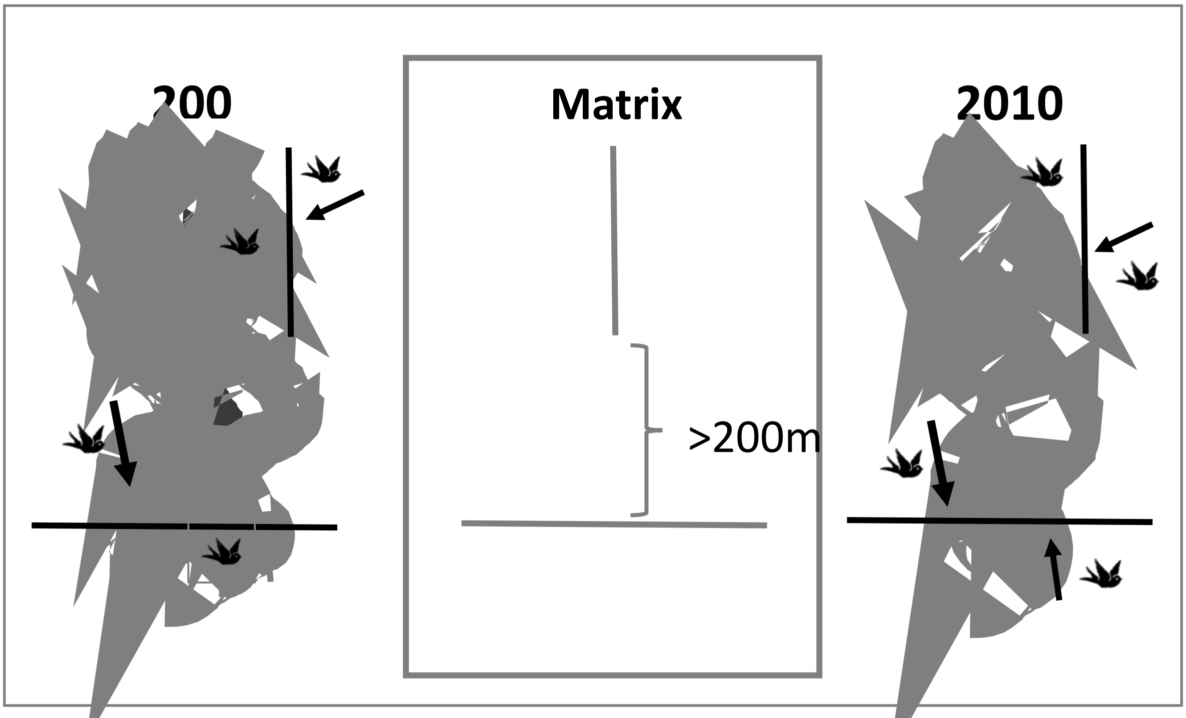

Figure 4.2 - Mist netting scheme used.

Nets remained at the same or approximately the same locations in both years. To avoid sampling bias nets were operated simultaneously during the sampling period and across habitat types. Nets were open 30 min before sunrise, were checked at one hour intervals and remained open 5 hours every morning during the sampling period, unless closing due to rain, high winds or temperature was mandatory. For each bird captured we recorded: location (net lane, habitat), time, ring number, ringer, species, fat score (Spencer1976), weight, age and sex (Svensson 1992). Bird species were classified according to their phenological categories in the south of Portugal in residents (R), short distance migrants (SDM) and long distance migrants (LDM) (Catry *et al*., 2010). Finally, species were grouped according habitat preferences in forest (F), bushes (B), aquatic (A) and ubiquist (U) (adapted from Roché 1986).

To validate if changes detected in our study could be attributed to wider population patterns or specifically to the habitat modifications in our sites, we used five days ringing data from 2007 and 2010 fall migration period (between 13th and 27th of September) provided by ”A Rocha” ringing station located in Mexilhoeira Grande, 25 km SW of our study sites. In “A Rocha” ringing station the same protocol, was used in both years. Ten mist nets were used: five 18 m nets, four 12 m nets and one 9 m net. Nets were erected in citrus groves and fields around “Quinta da Rocha” near Alvor River and estuary.

Data analysis

In order to compare data between years we standardized the bird captures by time per area ($h.m^2$), by dividing the daily captures by mist net area and the time it was open. We assumed that capture numbers are a function of bird abundance, which was indexed to the maximum number of individuals of each species captured standardized by $h.m^2$.

To evaluate the importance of riparian forest as an ecological corridor for songbirds, we performed non-parametric tests based in the following attributes: species richness (number of species captured in the first 100 captures and daily average number of species captured), species abundance (average number of birds captured daily) and body condition (fat score and weight). To compare habitat uses of the riparian forest and the matrix we used Kruskal-Wallis multiple comparisons tests, within the same year (for both years), for the different species, phenological categories and habitats, as well as for the different combinations of habitats and species, habitats and phenology categories. Bonferroni’s correction was used. To compare mean daily capture rates (diversity and abundance) after clear-cut we used Wilcoxon matched pair test for which we paired day one of 2007 with day one of 2010 and so on. We used two values for each day, corresponding to site one and two, generating a total of 26 observations per day for each year (2 sites x 13 days). We compared bird condition before and after clear-cut, fat and weight using Mann-Whitney test, which was also used to analyse if species richness and the number of days between captures were different after clear-cut. The software Statistica 10 (StatSoft Inc. 2011) was used in all statistical analysis.

4.4 Results

Control data

Species richness at "A Rocha" was the same in both years: 21 species in 2007 and in 2010. A total of 144 birds were captured in 2007 and 141 in 2010. No significant differences were observed between mean daily species diversity capture rates (z=0.135, p>0.05) or daily capture rates (*z*=2.70, p>0.05). Similarly, analysis by species did not detect significant differences between the numbers of birds captured daily in 2007 and 2010.

Species diversity and abundance

In Odelouca ringing sites, species richness was significantly higher before clear-cut than after (z=5.280, p<0.001). In all, 35 species were caught in both years and a total of 621 birds were captured (Table 4.2). The number of species captured in the first 100 captures was higher before (25) clear-cut than after (22). Three days were necessary to capture 100 birds in 2007, but six more days were necessary to capture the same amount in 2010. Mist nets caught significantly more birds (z=4.405, p<0.001, Wilcoxon's test for matched pairs) before clear-cut (Mean=0.66, SD=0.10) birds per ha/h.m^2 (n=25) than after (Mean=0.32, SD=0.39) birds per ha/h.m^2 (n=25).

Our results showed that the average number of species captured in riparian habitats (either the lateral or lengthwise dimensions) was higher than in upland (matrix) in both years. Before clear-cut the average number of species caught in the lengthwise dimension was significantly higher than the average number of species caught in lateral dimension (p<0.001, z=5.364) and upland (p<0.001, z=6.956). No differences were detected between the average number of species caught in lateral and upland. After clear-cut no differences were detected in the average number of species caught in the lengthwise and lateral dimensions. However, the average number of species captured in upland habitat was significantly lower compared with values obtained in lateral (p<0.01, z=3.253) and the lengthwise (p<0.001, z=3.962) dimensions.

More birds were captured in all habitats before clear-cut, with the lengthwise dimension showing the highest numbers (z=3.179, p<0.01), followed by lateral (z=2.062, p<0.05) and upland habitats (z=3.669, p<0.001). Before clear-cut the average number of birds caught in the lengthwise dimension was significantly higher than the average number caught in lateral dimension (p<0.001, z=5.402) followed by upland bushes (p<0.001, z=4.875). No differences were detected between the average number caught in lateral and upland bushes. After clear-cut no differences were detected in the average number caught between upland bushes and both riparian dimensions. However, the average number captured in lengthwise dimension remained significantly higher compared to the average number caught in lateral dimension (p<0.05, z=2.58).

Bird condition, age and recapture analysis

In Odelouca four bird species were significantly heavier prior to clear-cut than after, while two species were heavier after clear-cut (Table 4.3). Garden Warbler (*Sylvia borin*), a LDM was fatter before the clear-cut, and Eurasian Blackcap (*Sylvia atricapilla*), a resident species was fatter after clear-cut (Table 4). Resident birds were significantly younger after clear-cut (Mean= 2.86, SD=0.7) then before (Mean=3.2, SD=0.7) (z= 2.89, p<0.001). The analysis of recaptured birds, revealed that the number of days between captures for migrant birds was (*z*=-2.123, p<0.05) larger before (Mean=3.7, SD=3.3 days, n=21) than after clear-cut (Mean=1.4, SD=1.7 days, n=7) but not for residents (*Z*=1.308, p>0.05).

In “A Rocha” none of the species that showed a clear condition decline in Odelouca had a significant decline either in weight or fat; nevertheless, Reed Warbler (*Acrocephalus scirpaceus*) showed a significant decrease in fat and weight (Table 4.3). There were no differences between 2007 and 2010 in the age of resident and migrant birds.

Table 4.2. Bird species captured in mist-nets at River Odelouca before (2007) and after (2010) clear-cut operations;

	Longitudinal dimension			Lateral dimension			Upland				
Species[a]	Before	After	D[b]	Before	After	D	Before	After	D	Phenology[c]	Habitat[d]
Acrocephalus schoenobaenus	0	3.5SD0.1 (1)	+	0	0	0	0	0	0	LDM	A
Acrocephalus scirpaceus	42.1SD0.6 (9)	9.8SD1.4 (3)	-	0	3.7SD0.3 (4)	+	0	0	0	LDM	A
Aegithalos caudatus	61.8SD0.9 (13)	39.2SD1.0 (9)	-	2.8SD0.1 (1)	19.5SD0.3(8)	+	0	15.0SD0.4 (2)	+	R	B
Alcedo atthis*	**74.7SD4.4 (16)**	**16.7SD1.5 (4)***	**-**	5.7SD1.1 (2)	0	-	0	0	0	R	A
Carduelis chloris	0	0	0	13.8SD0.1 (5)	0	-	0	0	0	R	B
Certhia brachydactyla *	**38.6SD2.2 (8)**	**0***	-	5.7SD0.7 (2)	0	-	0	0	0	R	F
Cettia cetti **	**81.1SD3.5 (17)**	**3.4SD0.6 (1)****	-	2.5SD0.5 (1)	1.8SD0.3 (1)	-	5.3SD1.0 (1)	0	-	R	B
Cyanopica cyanus	13.2SD1.9 (3)	0	-	0	0	0	0	0	0	R	F
Emberiza cia	28.6SD2.7 (6)	0	-	2.8SD0.5 (1)	10.9SD0.0(5)	+	15.5SD 1.6(3)	9.3SD 1.2(2)	-	R	B
Emberiza cirlus	18.1SD2.7 (4)	0	-	5.7SD1.1 (2)	0	-	0	0	0	R	B
Erithacus rubecula **	**156.1SD6.1 (32)**	**25.7SD2.5 (6)****	-	**60.4SD 3.9 (21)**	**14.2SD 1.4(6)***	**-**	31.7SD 2.7 (6)	12.1SD 1.7(2)	-	SDM	F
Estrilda astrild	0	19.4SD2.1 (4)	+	0	0	0	0	0	0	R	B
Ficedula hypoleuca **	**63.9SD3.1 (13)**	**3.4SD0.6 (1)****	-	**30.0SD1.8 (11)**	**3.3SD0.6 (1)***	**-**	10.7SD1.5 (2)	4.6SD0.9 (1)	-	LDM	F
Fringilla coelebs	0	0	0	5.7SD0.8 (2)	0	-	0	0	0	R	F
Garrulus glandarius	9.9SD1.3 (2)	0	-	2.5SD0.5 (1)	0	-	5.3SD1.0 (1)	0	-	R	F
Hippolais polyglotta	0	0	0	2.8SD0.5(1)	0	-	4.7SD0.9 (1)	0	-	LDM	B
Locustella naevia	4.9SD 1.0(1)	0	-	5.7SD0.8 (2)	0	-	10.7SD 2.1(2)	0	-	LDM	A
Motacilla cinerea	4.4SD0.9 (1)	24.1SD2.5 (5)	+	0	0	0	0	0	0	R	B
Muscicapa striata	0	0	0	2.3SD0.4 (1)	0	-	0	0	0	LDM	F
Cyanistes caeruleus*	**34.2SD2.6 (7)**	**5.7SD1.1 (1)***	-	**19.3SD1.7 (7)**	**0***	-	16.1SD1.7 (3)	3.9SD0.8 (1)	-	R	F
Parus major	**29.7SD2.5 (6)**	**0***	**-**	11.5SD1.8 (4)	8.2SD1.1 (3)	-	0	0	0	R	F
Passer domesticus	19.2SD2.2 (4)	0	-	0	0	0	0	0	0	R	U
Phoenicurus phoenicurus	9.9SD1.3 (2)	0	-	8.3SD0.9 (3)	2.2SD0.4 (1)	-	0	6.9SD1.4 (1)	+	LDM	B
Phylloscopus collybita	4.9SD1.0 (1)	0	-	2.8SD0.6 (1)	0	-	0	0	0	SDM	B
Phylloscopus trochilus	46.5SD3.7 (10)	42.1SD3.8 (7)	**-**	**2.8SD0.6 (1)**	**13.6SD1.0 (6)***	**+**	0	19.2SD2.3 (3)	+	LDM	B
Picus viridis	4.4SD0,9 (1)	0	-	0	0	0	0	0	0	R	F
Regulus ignicapilla	0	5.7SD1.1 (1)	+	2.8SD0.6 (1)	0	-	0	0	0	SDM	F
Saxicola torquata	0	7.6SD1.0 (2)	+	8.6SD1.2 (3)	5.8SD0.6 (3)	-	15.5SD2.2 (3)	0	-	R	B
Sturnus unicolor	8.8SD1.7 (2)	0	-	0	0	0	0	0	0	R	B
Sylvia atricapilla **	**303.9SD11.0 (63)**	**59.5SD4.0 (18)****	**-**	84.0SD4.3 (30)	72.6SD5.5 (27)	**-**	**37.0SD2.8 (7)**	**0***	**-**	SDM	B
Sylvia borin	**59.5SD 4.0 (12)**	**16.1SD 1.5 (4)***	**-**	5.5SD 0.7 (2)	9.9SD 1.1 (3)	+	0	0	0	LDM	B
Sylvia cantillans	0	0	0	2.3SD0.4 (1)	2.0SD0.4 (1)	-	10.1SD1.4 (2)	0	-	LDM	B
Sylvia communis	9.9SD1.3 (2)	15.8SD1.9 (3)	+	2.3SD0.4 (1)	6.1SD0.9 (3)	+	0	0	0	LDM	B
Sylvia melanocephala **	49.0SD3.7 (10)	9.6SD1.3 (2)	-	36.7SD2.3 (13)	22.7SD1.5 (10)	**-**	**99.2SD5.4 (19)**	**15.4SD1.7 (3)****	-	R	B
Sylvia undata	0	10.1SD1.4 (2)	+	0	4.9SD1.0 (1)	+	10.1SD1.4 (2)	0	-	R	B
Troglodytes troglodytes *	19.3SD1.8 (4)	0	-	11.6SD1.3 (4)	1.8SD0.4 (1)	-	4.3SD0.8 (1)	3.8SD0.8 (1)	-	R	F
Turdus merula *	**52.9SD3.1 (11)**	**7.3SD1.0 (2)****	-	14.5SD1.8 (5)	4.0SD0.5 (2)	-	10.2SD1.4 (2)	0	-	R	F
Number of species	27	19		28	18		15	9			
Sampling effort – m (nets used)	36 (2*18)	24 (2*12)		21 (12+9)	42 (18+15+9)		39 (15+2*12)	48 (4*12)			

Footnote: a) Species captured in each sampling season in all habitats are denoted by the mean birds captured by m^{2}h - (meanSD)*10^{-3}; b) Differences between capture daily capture rates: positive (+), negative (-) or neutral (0); c) Resident (R), short distance migrant (SDM) and long distance migrant (LDM) (adapted from Catry et al. 2010). Significant statistical differences between years are in bold.*p<0.05, **p<0.01, *** p<0.001 indicate p-level results of Wilcoxon matched pair test; d) Habitat preferences are: forest (F), bushes (B), aquatic (A) and ubiquist (U) (adapted from Roché 198

Table 4.3. Bird condition in Odelouca River and “A Rocha” ringing sites: weight and fat score before (2007) and after clear-cut (2010).

	Odelouca River			“A Rocha” (control site)		
	Weight before[a]	Weight after[a]		Weight before[a]	Weight after[a]	
Erithacus rubecula	15.4SD1.0 (51)	13.7SD4.0 (4)	z=2.845, p<0.01	15.2SD0.1 (4)	-	p>0.05
Sylvia melanocephala	11.9SD0.7 (41)	11.4SD0.5 (16)	z=2.547, p<0.01	-	11.4SD0.6 (3)	p>0.05
Fycedula hypoleuca	14.1SD1.0 (29)	12.9SD0.7 (3)	z=1.747, p=0.08	15.1SD1.0 (14)	14.5SD1.6 (20)	p>0.05
Sylvia borin	20.9SD3.7 (11)	17.4SD1.7 (7)	z=2.0.83, p<0.05	22.9SD3.4 (21)	21.5SD3.0 (25)	p>0.05
Alcedo atthis	33.0SD1.3 (19)	34.7SD1.4 (4)	z=-1.950, p<0.05	31.1 (1)	-	p>0.05
Cyanistes caeruleus	7.9SD2.5 (13)	9.9SD0.7 (2)	z=-1.964, p<0.01	-	-	p>0.05
Acrocephalus scirpaceus			p>0.05	14.3SD2.2 (5)	11.4SD1.5 (15)	z=2.358, p<0.05
	Fat score before[b]	Fat score after[b]		Fat score before[b]	Fat score after[b]	
Sylvia atricapilla (SDM)	0.7SD1.1 (85)	1.4SD1.4 (47)	z=-2.715, p=0.012	4.1SD1.6 (8)	5 (1)	z=1.0, p>0.05
Sylvia borin (LDM)	3.6SD1.7 (11)	1.6SD1.4 (7)	z=-2.960, p<0.01	4.8SD2.0 (21)	4.7SD1.5 (25)	z=0.644, p>0.05
Acrocephalus scirpaceus (LDM)				6SD1.7 (5)	2.7SD2.1 (14)	z=2.567, p<0.001

a) Weight before/after: meanSD (n); Fat score before/after: meanSD (n)

4.5 Discussion

Species diversity and abundance

Our results are in line with studies that have documented high diversity and abundance of birds in Mediterranean riparian ecosystems (Décamps & Décamps, 2002; Blondel, 2003; Godinho *et al*., 2010; Pereira *et al*., 2014). The higher bird diversity and abundance recorded in riparian forest than in upland habitats (matrix) in both years reflects a similar pattern observed in other geography areas (LaRue *et al*., 1995; Bentley & Catterall, 1997; Mosley *et al*., 2006) where a clear water availability contrast exists between riparian habitat and upland. Although this result alone does not show that birds used the riparian forest to move between patches, it indicates that in Mediterranean xerophilous matrices, riparian forests support more diverse communities than the matrix, thus reinforcing their conservation value as habitat corridor.

We are aware that most studies that inferred corridor usage by comparing species diversity and abundance in connected and unconnected patches could not directly infer movement in corridors (Rosenberg *et al*., 1996), because the presence of a species is not an indicator of movement use. Our survey method allowed us to determine bird direction when captured in the riparian area and therefore we were able to infer bird use of the riparian forest as ecological corridor (through nets placed perpendicular to the river) and as interface habitat (through nets placed parallel to the river). *Mosley et* *al*. (2006) made similar inferences although with a slightly different net scheme indicating that riparian areas may function as movement corridors for birds.

A meta-analytical review made by Gilbert-Norton *et al*. (2010) found that corridors were more important for the movement of invertebrates, non-avian vertebrates, and plants than they were for birds. Cushman *et al.* (2013) mention that birds may be able to fly over sections of matrix habitat and as a result are less likely to move through corridors than non-avian vertebrates. However, our study reveals statistically significant evidence that riparian areas are used as movement corridors since lengthwise dimension captured higher bird rates than lateral dimension. Similar results were found by Mosley *et al.* (2006) in Canada during the fall migration period.

Considering that we have found more migratory birds in this lengthwise dimension, it is likely that they use riparian areas as corridors because these habitats show a lower resistance values to movement then the matrix. These findings are of great applicability to conservation since landscape resistance to movement is the first step to design biological corridors (Rosenberg *et al*., 1995; Cushman et al., 2013). Following Rosenberg *et al*. (1995) definition, we showed that *immigration to the target patch via corridor* (that is riparian habitat) *is greater than if the corridor were absent* (after clear-cut). The role of river corridors as stopover habitats for migratory birds on their route to African quarters was explored by Catry *et al*. (2009) for diurnal migrants crossing dry sectors of SW Iberia. Although they did not found a significant migratory corridor along the lower Guadiana River during fall migration, more studies are needed specially focusing on the role of riparian habitats as stopover sites.

Abundance of SDM suffered huge decrease in all habitats particularly due to declines of Eurasian Blackcap and Robin (*Erithacus rubecula*) indicating that available habitat no longer provides sufficient energy to support these birds. This result is particularly important if we consider that migrating birds need to cope with annual habitat variation along known migrating routes that may cause even higher mortality rates during a period known to be the most critical time of the annual cycle (Carlisle *et al*., 2010) and that they show site-fidelity during migration (Cantos & Tellería, 1994; Mettke-Hofmann & Gwinner, 2003).

Movements to suitable habitat are likely to have occurred, which is supported by the fact that after clear-cut the time between re-captures is smaller and the proportion of juveniles is higher (similar results were obtained by Machtans *et al*., 1997).

The number of days migrant birds remained in the area is smaller after riparian forest removal and consistent with what was hypothesized by Rosenberg *et al*. (1997) which stated that if animals "do select low-quality corridor, they will move through it more quickly". These findings reinforce the need to provide alternative short distance suitable habitat whenever a massive clear-cut is necessary.

Lourenço *et al*. (2010) refer that birds may not remain at sites as long when less foraging habitat and fewer resources are available. After clear-cut some habitat characteristics between depleted riparian forests and the matrix become less pronounced (*e.g*. bush height and density, presence of trees) and may have increased the use of riparian habitats by bush like species due to lack of suitable habitat in the upland bushes. In fact, a shift between habitats surveyed seems to have occurred at least for some species: the majority of Eurasian Blackcaps records, prior to riparian forest removal, are on the lengthwise dimension, but after clear-cut, lateral dimension captured more birds of this species, indicating that the poor quality riparian forest does not provide all the resources that migratory birds need, leading to wider movements between riparian habitat and the matrix. On the contrary, for two resident species, Sardinian Warbler (*Sylvia melanocephala*) and Dartford Warbler (*Sylvia undata*), the majority of birds captured prior to riparian forest removal was in the upland bushes, but due to changes in the matrix the majority of records after clear-cut was in lateral dimension. Displacement from upland habitats to riparian forest may have occurred and thus support the idea that in depleted environments species may move to nearby locations with similar habitat characteristics (Rosenberg *et al*., 1997). These findings indicate that depleted riparian forest still plays an important role at the landscape level at least for some species (Skagen et al., 1998) and are consistent with interface use and the influence that upland bird communities make in composition within a riparian site and vice versa (Szaro & Jackle, 1985; Knoff & Samson, 1994; Pereira *et al*., 2012). The Kingfisher (*Alcedo atthis*) showed a drastic decrease probably because of the known impact that forest removal has on fish assemblages (Dale Jones III *et al*., 1999) and maybe also due to decrease on availability of perching positions for fishing. Our results demonstrate that riparian forest removal greatly affects bird communities, indicating a general deterioration of suitable habitat for many bird species as a result of clear-cut. Factors both at landscape level (major habitat deforestation) and in-stream habitat level (decrease in macro-invertebrate abundance and diversity) may have influenced bird occurrence.

We believe that differences in the capture rates detected between habitats and years are reliable indicators of changes in bird assemblages, because capture rates were standardized. Moreover, the total net area used after clear-cut was higher and we still detected decreases in bird diversity and abundance. In the absence of a "true" control site (*e.g*. in the same river and with similar habitats) due to logistic constrains, we used data for the exact time frame from the nearest ringing station – "A Rocha". Captures from the control site did not showed differences between years, dismissing wider population trends and associating the Odelouca changes to habitat removal. Bird abundance and richness in Odelouca ringing sites shows a marked decline between years, with a decrease of 50% of bird abundance, due to a response to habitat change following clear-cut.

Bird condition, age and recapture analysis

To our knowledge this is the first time that an impact on bird condition (weigth and fat scores) due to riparian forest removal has been reported, though Green *et al*. (2011) have found two species which body mass declined as reservoir water levels increased, but direct influence of riparian quality in body condition could not be established. Burton *et al*. (2006) have reported impact on redshank (*Tringa totanus*) condition (weight decrease) due to habitat loss (dam filling – habitat un-availability).

Results presented in Table 4.3 suggest that poor quality corridors might function as ecological traps (Schlaepfer *et al*., 2002) at least for some species, which should be considered when designing mitigation measures in riparian corridors. Care should be taken since some characteristics of riparian forest (*e.g.* continuity and heterogeneity) may not be able to provide birds the needed resources for survival during migration (Finch & Young, 2000) when energy demands are greater and mortality rates are probably high (Green *et al*., 2011). However, it is important to mention that weigh is influence by size.

While Eurasian Blackcap fat scores significantly increased after clear-cut Garden Warbler levels decreased. The differences in these two insectivorous birds that are frugivorous during migration can tentatively be explained by their different ecological strategies during migration through Iberia: Eurasian Blackcap ingests less proportion of fruits and higher proportion of insects, whereas Garden Warbler relies greatly on fleshy fruit (Jordano, 1988). After the clear-cut, when scrub and fruit available have been greatly reduced, Eurasian Blackcaps might have changed to a more insectivorous diet and because of that birds became fatter, since insects have been demonstrated to be responsible for higher scores of fat and weight (Jordano, 1988). In the absence of scrubs which produce fleshy fruit, Garden Warbler fat deposits become depleted. We cannot exclude the fact that Garden Warblers are LDM coping with an unpredictable habitat change in a migration route, while Eurasian Blackcaps are SDM that winter in the area and thus may know and explore local landscapes in a more efficient way.

Implications for management and policy

Although we did not compare fragments of connected versus unconnected habitat, we did analyse the diversity and abundance of bird species of a linear habitat type, the riparian forest, and the matrix, and compared its use in a mature versus depleted situation. The results clearly demonstrate that after riparian forest removal species diversity and abundance decrease, the resistance to movement of the habitat matrix and riparian forest is different (even after clear-cut), a shift in the habitat use occurred and impact in bird condition (weight and fat) of some species also occurs. Our results support the idea that riparian habitats do function has ecological corridors for songbirds.

These findings are particularly relevant in the Mediterranean Region since habitat destruction and fragmentation together with climate change may be responsible for high extinction rates of species occurring in this region (Cuttelo *et al*., 2008). In fact, mature Mediterranean riparian forests are linear oases of vegetation on semi-arid landscapes where birds can rest and refuel when traveling. As a result the development of networks of ecological corridors is even more important. Despite the demonstrated impact on migrating songbirds, the impact on sedentary birds should not be neglected since they are five times more likely to go extinct from climate change than long-distance migrants (Sekercioglu *et al*., 2008).

We must conclude that irrespective of the corridor definition that we consider, that is habitat linked definition (Forman & Godron, 1986) or conduit definition (Simberloff *et al*., 1992), we were able to demonstrate that riparian songbird assemblages are different from the matrix, and that immigration to the target patch via riparian forest was greater in its presence than if the riparian forest was absent (Rosenberg *et al*., 1995 hypothesis). We support Hess & Fisher (2001) who suggested that conservationists and planners should consider all possible functions of corridors (habitat, conduit, filter, barrier, source, sink/trap) when designing them. It is important to take into account the possibility of negative, unintended consequences of corridor creation in their design. In the same way that corridors may facilitate movement of rare, endangered, or declining species, they may also increase mortality rates by not being able to provide sufficient energy for survival, particularly for migrant species. This is particularly important since migrant birds seem to have evolved special cognitive abilities that enable them to return to the same breeding, wintering, and stopover sites in successive years (Mettke-Hofmann & Gwinner, 2003) and site-fidelity during migration has been reported in Iberian Peninsula for three species of warblers including blackcap (Cantos & Tellería, 1994). So, it is important to conserve high-quality stopover sites, where birds can rapidly accumulate energy (Green *et al*., 2011). Skagen *et al*., (1998) advocate that all riparian patches (in southeastern Arizona) are important as stopover sites to migratory birds regardless of their size and degree of isolation. Our results show that this may not stand for all species and all phenological categories in other parts of the world. Work is still needed to understand when and how riparian habitats may function as habitat corridors and which habitat characteristics favor higher fitness of bird assemblages without causing negative effects, especially because impact on bird condition due to habitat changes has been detected elsewhere (Burton *et al*., 2006).

The results are also of importance at European level because common European bird species are declining (Inger *et al*., 2015) and SW Iberian Peninsula, is in the East Atlantic Flyway, which is an important area for trans-Saharan migrants known as the last stronghold before crossing the sea on their way to Africa. Therefore, an articulation of Water Framework Directive, Habitats Directive and Birds Directive are imperative to achieve multipurpose goals and broader conservation strategies involving "out of countries perspective" and integrate all animals' lifecycles habitats. Since rivers are the only natural linear continuous landscape units they may be used to contribute to the development of networks of ecological corridors. Additionally, it is important to mention that riparian zones have acted as safe sites for regional flora during dry periods (*e.g*. Pleistocene droughts) (Naiman & Décamps, 1997), thus riparian forests restoration for multi-purposes can be important for promoting ecological connectivity in a changing climate (Krosby *et al*., 2010).

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Chapter 5

EVALUATING THE RESPONSE OF BIOLOGICAL ASSEMBLAGES AS POTENTIAL INDICATORS FOR RESTORATION MEASURES IN AN INTERMINTENT MEDITERRANEAN RIVER

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5. Evaluating the Response of Biological Assemblages as Potential Indicators for Restoration Measures in an Intermittent Mediterranean River

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Key words: River restoration, Benthic macroinvertebrates, Fish, Birds, Macrophytes, Environmental gradientes, Pressure gradientes, Multivariate analyses, Concordance

5.1 Abstract

Bioindicators are essential for detecting environmental degradation and for assessing the success of river restoration initiatives. River restoration projects require the identification of environmental and pressure gradients that affect the river system under study and the selection of suitable indicators to assess habitat quality before, during and after restoration. We assessed the response of benthic macroinvertebrates, fish, bird and macrophyte assemblages to environmental and pressure gradients from sites situated upstream and downstream of a cofferdam on the River Odelouca, an intermittent Mediterranean river in southwest Portugal. The Odelouca will be permanently dammed in 2010. Principal Component Analyses (PCA) of environmental and pressure variables revealed that most variance was explained by environmental factors that clearly separated sites upstream and downstream of the partially built cofferdam. The pressure gradient describing physical impacts to the banks and channel as a result of land use change and groups of clusters were less distinct than those formed from environmental data. Redundancy Analysis revealed significant levels of explained variance to species distribution patterns in relation to environmental and pressure variables for all 4 biological assemblages. Partial Redundancy analyses revealed high levels of redundancy for pH between groups and that the avifauna was best associated with pressures acting upon the system. Patterns in invertebrates and fish were associated with descriptors of habitat quality, although fish distribution patterns were affected by reduced connectivity. Procrustean and RELATE (Mantel test) analyses gave broadly similar results and supported these results. In the light of our findings, we give suggestions on the suitability of key indicator groups such as benthic macroinvertebrates and endemic fish species to assess in stream habitat quality and appropriate restoration measures, such as the release of peak flow patterns that mimic intermittent Mediterranean systems to combat habitat fragmentation and reduced connectivity.

5.2 Introduction

Bioindicators have long been used in ecological assessment of surface water quality, which is subject to degradation from impacts ranging from agriculture to industrialisation and urbanisation (Heino and others 2002; Hughes 2005; Karr 1999). They are also used to gauge the success of restoration and restoration measures (Gore and others 2001; Kondolf 1995a; Kondolf 1998). Common indicators of lotic condition include benthic macroinvertebrates (Bonada and others 2006; Feld and Hering 2007;

Heino 2005; Rundle and others 1992), fish (Godinho and others 2000; Hughes and others 2005; Karr 1981), diatoms (Kelly and others 1998; Round 1991) and macrophytes (Dodkins and others 2005; Ferreira and Aguiar 2006; Ferreira and others 2002). Birds have also been used to a lesser extent to assess the wider riverine landscape such as the riparian gallery (Bryce and others 2002; Jansen and Robertson 2001b; Vaughn and others 2007). To assess ecosystem health, an effective bioindicator should exhibit detectable and measurable levels of change in relation to specific environmental or pressure gradients, ideally starting from reference conditions (Johnson and others 2006a; Karr and Chu 2000; Paavola and others 2006). In this paper we use the word “pressure” to describe sources of degradation or impacts upon river systems and the word “concordance” to describe the degree of association between changes in taxonomic composition in a biological assemblage and environmental or pressure gradients (Paavola and others 2006; Sánchez-Montoya and others 2007).

The European Water Framework Directive (WFD) (European Commission 2000) and the development of lotic monitoring systems in the USA (Barbour and others 1999, Hughes and Peck 2008, Paulsen and others 2008, Stoddard and others 2008) have led to considerable growth in the number of monitoring methods based on the assemblages mentioned above. Bioassessment or restoration measures must consider fundamental changes in aquatic assemblages in relation to environmental quality to assess the success of a restoration or rehabilitation efforts (Kondolf 1995a). Assessment systems that use several biological assemblages (O'Connor and others 2000) consider the concordance of elements of the assemblages with aspects of river environment health to provide a comprehensive ecological image of river health, a concept that is central to the WFD. However, high levels of redundancy can occur in multi assemblage monitoring programmes and relevant information on ecological status can be derived from few or even a single biological assemblage (Resh 2008).

Mediterranean rivers are highly distinct systems, with remarkable but predictable natural cycles of flood and drought that vary in intensity according to levels of annual and interannual rainfall (Bêche and Resh 2007b; Gasith and Resh 1999; Pires and others 2004). This powerful environmental filter determines biological community traits (Bonada and others 2005) and is a strong source of covariance which can detrimentally affect the detection of anthropogenic impacts by biological assemblages. Impacts on Iberian Mediterranean rivers include habitat fragmentation, reduced lateral and longitudinal connectivity and destruction of natural flow regimes as a result of agriculture, forestry, damming, water abstraction and urbanisation (Aguiar and Ferreira 2005; Bonada and others 2005; Hughes and others 2008; Poff 1997).

The Odelouca is an intermittent Mediterranean river (Algarve region, southern Portugal) that, despite human intervention, has intact and floristically diverse riparian galleries along considerable stretches of the river corridor. Two critically endangered endemic fish species (Pires and others 2004) also occur in the Odelouca, namely *Iberochondrostoma almacai* (Coelho and others 2005) and *Squalius aradensis* (Coelho and others 1998). Ongoing government initiatives to improve water supply in the Algarve region have authorised completion of a partially constructed dam on the Odelouca by 2010; however compulsory environmental mitigation and compensation measures have been implemented to offset impacts caused by the dam's construction.

This paper assesses the response of four biological assemblages (benthic macroinvertebrates, fish, birds and macrophytes) to environmental and pressure variables in order to identify suitable indicators for monitoring the progress of restoration measures. We aim to answer the following questions: (1) what are the principal environmental and pressure gradients acting upon the Odelouca and how does each biological assemblage respond to them? (2) Does a single group best describe changes in the system under study or is an integrated approach better for assessing restoration measures? (3) Which restoration measures are recommended for the Odelouca?

5.3 Methods

Study Area

The Odelouca River (catchment area 511.4 km^2) is a medium-sized, low-gradient, intermittent lowland stream running through predominantly schistose areas typical of southern Portugal (Figure 5.1). The area's Mediterranean climate exhibits a predictable seasonal pattern of rainfall (wet season from October to March, dry season from June to September). The Odelouca is relatively slow running, subject to "flashy" spates in the winter, diminishing to unconnected, temporary pools in the dry riverbed during the summer. Catchment topography varies from narrow steep sided valley walls to restricted meander valleys and small floodplains. Greater habitat heterogeneity occurs in less disturbed mid sections of the river corridor and tributaries (temporary side channels, backwaters and riparian galleries). Woody riparian plants comprise *Alnus glutinosa* (L.) Gaertner, *Salix salviifolia* Brot. ssp.

australis Franco, *Nerium oleander* L, and *Fraxinus angustifolia* Vahl. Stands of *Tamarix africana* Poiret and *Nerium oleander* L. occur in the lower reaches of the basin.

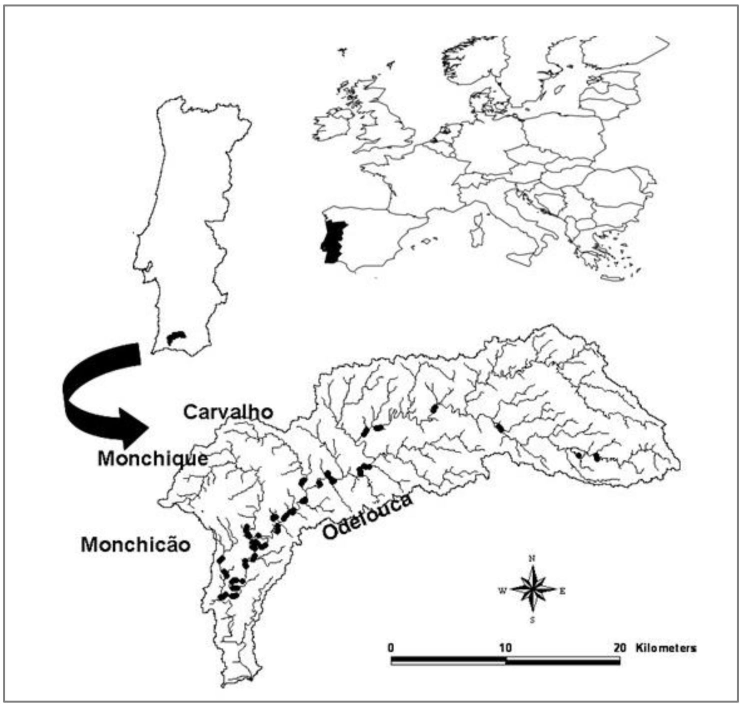


Figure 5.1 – Map showing where Portugal is situated and the Odelouca basin with sampling sites. The thick black bar indicates where the cofferdam is situated.

Agriculture (extensive citrus groves and low level grazing) has replaced the natural Mediterranean cork-oak woodland vegetation (*Quercus suber* L.) on the wider floodplain below the partially built dam (a cofferdam and an excavated subterranean flow diversion gallery). Observed impacts include diffuse organic pollution, nutrient enrichment, riparian clearance, bank resectioning, bank reinforcement and reduced longitudinal connectivity from irrigation (pumping from the riverbed and small scale damming). Urbanisation is scant, restricted to two small villages and small agricultural hamlets. *Eucalyptus globulus* Labill. and *Pinus pinaster* Aiton plantations are present in parts of the upper basin. Tributaries suffer little physical disturbance but the Monchique stream is affected by organic input from piggeries and Monchique village. The lower reaches of the Monchicão tributary are affected by abstraction for agricultural irrigation. The Ribeira de Carvalho, situated in the upper Odelouca catchment, is far less altered.

Habitat Assessment

Field data were collected in spring 2005 from 30 sites: 25 sites along the main channel and 5 sites along the tributaries of Ribeira de Carvalho (1 site), Ribeira de Monchique (2 sites) and The Ribeira de Monchicão (2 sites), near their confluences with the Odelouca. Habitat structure, diversity and quality were assessed over a 500m reach using an adapted version of the River Habitat Survey (RHS; addition of Iberian Peninsula land use categories and plant species). Developed in the UK, RHS records in-stream substrate and flow type, character and modification of the margins, land use, riparian vegetation, predominant habitat features and modifications, together with measurements of stream and bank dimension (Raven and others 1997).

A geographical information system of land use, riparian vegetation quality, conservation state and continuity was created from survey data and aerial photography. Catchment data on geology, climate, altitude, relief, land use, land cover, organic and industrial discharge and the presence of roads were also added (Fernandes and others 2007).

Lotic communities

All organism group samples were taken within each 500m RHS reach. Aquatic organisms were sampled using European STAR project methodologies, modified for application in Portuguese lotic systems by the Portuguese Water Institute INAG (http://dqa.inag.pt/dqa2002/port/docs_apoio/nacionais.html). A full taxon list and species codes are given in Appendix 1.

Benthic macroinvertebrate samples were taken along a 50-100m long stretch at the downstream end of each 500m reach using an adapted AQEM (Integrated **A**ssessment System for the Ecological **Q**uality of Streams and Rivers throughout **E**urope using Benthic **M**acroinvertebrates) multihabitat sampling protocol (AQEM 1999; Hering and others 2004; INAG 2008a). The sampling area covered the greatest possible diversity of habitats in the reach, including (if present) a riffle and areas of deposition. Types and extent of habitats (organic and inorganic substrates)were visually estimated prior to sampling and six 1m long 25cm wide sampling units of the most representative habitats were taken (0.25m x 0.25m handnet; 500 µm mesh, habitats representing < 5% total cover were excluded). Samples, which were not timed, were proportionally distributed among the identified habitat types. The composite sample was placed in a labelled plastic flask and fixed *in situ* using 4% formaldehyde. In the laboratory, samples were washed, sieved, sorted and identified using a low-power stereo microscope. All individuals were picked from the samples; sub-sampling was used when more than 200 individuals of a given taxon were present in the sample. Macroinvertebrates were identified to the lowest possible taxonomic level, which was mostly genus or species. Where higher levels of taxonomic resolution were not possible (for example many Diptera, or early instars of some insect groups) the AQEM protocol for taxonomic adjustment was applied (Hering and others 2002).

Fish were sampled using an adapted STAR methodology (INAG 2008b) with an Electracatch International, SAREL model WFC7-HV electrofisher (applying 300V and maintaining a 3-A output to a 40cm diameter ring anode). Sampling was carried out (area sampled was 20 times the mean width of the survey reach, minimum length 100m starting at spot check 1) by walking upstream in a zigzag pattern or sampling from a boat at sites over 1m deep (Godinho and others 2000). The same person operated the electrofisher across sample sites and two netters picked up stunned fish. Captured fish were held in large plastic containers, identified to species, counted, and immediately returned to the river. No voucher specimens were taken.

A trained field ornithologist carried out bird surveys over eleven days between late May and June to ensure the maximum number of nesting species. Surveys were not carried out in wet or windy conditions. Bird samples were made at three equidistant points (250m distance between points) including the riparian gallery and surrounding area (Bibby and others 2000). All birds observed or heard were recorded to species over a 10 minute period at each point. Records started as soon as observer got into the site, birds leaving the area where recorded as well as birds entering. The distance between the bird and the observer was estimated when the distance was inferior to riparian gallery width. This procedure allowed verification of observer efficacy so that bias was not introduced into the data set. Birds were also surveyed on the flood plain approximately 100m perpendicular to each point in the riparian gallery.

Macrophyte inventories (INAG 2008c) were carried out along 100m long reaches starting at the beginning of the 500m RHS reach, estimating percentage cover of each species in the sample area (minimum percent cover included 0.1%). Species level determinations were made on site; specimens that could not be identified were taken to the Superior Agronomy Institute herbarium (Technical University of Lisbon) and bryophytes to the herbarium of the Botanical Garden (Science Faculty of Lisbon) for identification (epiphytic bryophytes above the splash/humid zone were not included in the surveys).

Data analyses

Redundant environmental and pressure parameters were removed using the Spearman Rank Correlation analysis method (Feld and Hering 2007; Hughes and others 2009). If two environmental or pressure variables under comparison were highly correlated (threshold value of $r \geq 0.5$ or $r \leq -0.5$) the variable with the higher mean correlation coefficient was excluded from further analysis.

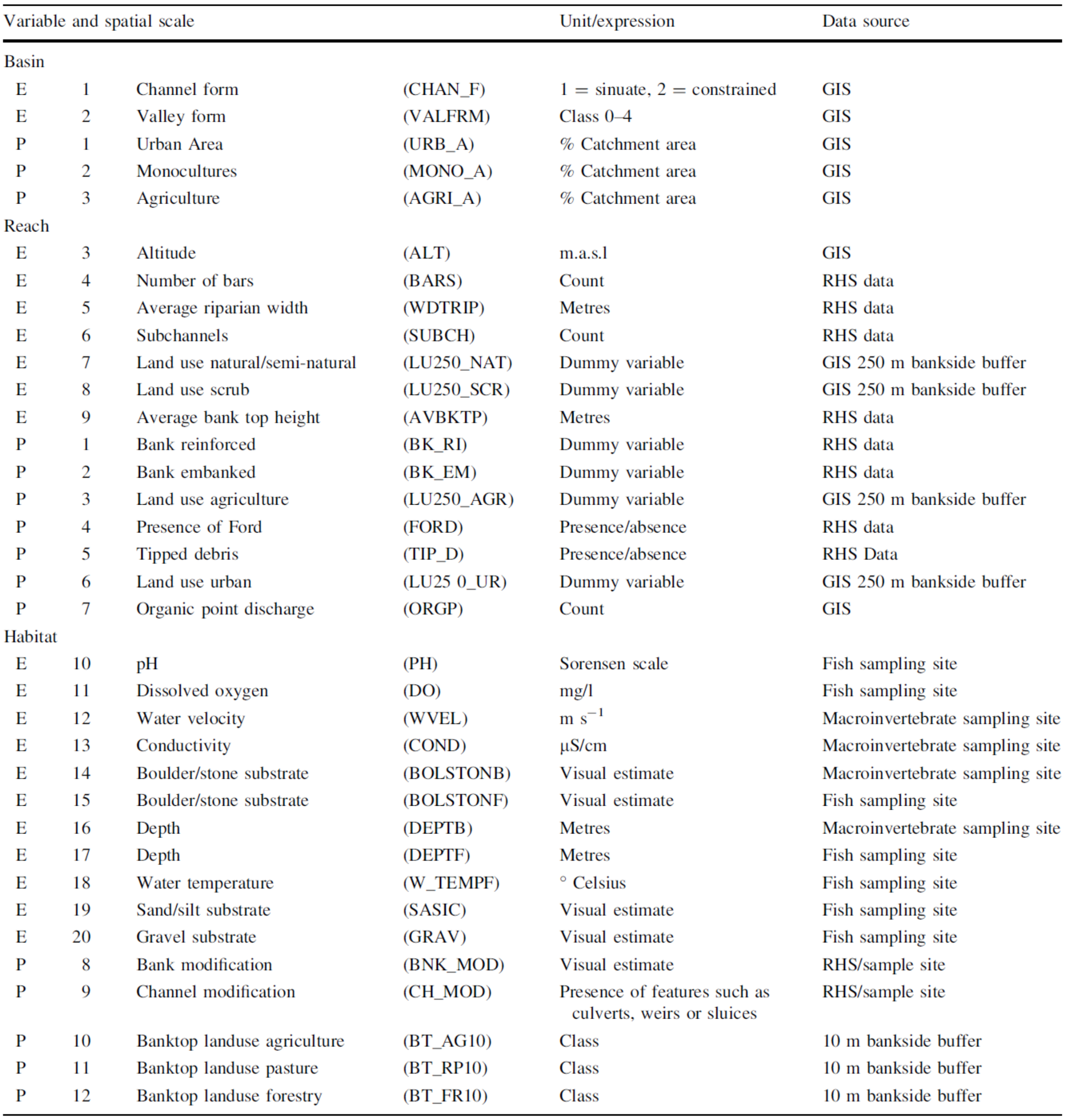
Principal Components Analyses (PCA) were carried out on retained log (x+1) transformed environmental and pressure data (binary data and ranked variables were not transformed) to reduce data dimensionality and identify the principal environmental and pressure gradients. From an initial list of 38 environmental variables and 36 pressure variables, a total of 20 environmental and 12 pressure variables were retained (Table 5.1). The loading value of retained variables (see Table 5.2) provided an indication of the proportion of their variance with a given PCA component and thus its contribution to distribution patterns in the ordination space. Taxonomic data for each organism group were log (x+1) transformed (singly occurring specimens and those occurring at fewer than three sampling sites excluded from further analyses) were analysed via Detrended Correspondence Analysis (DCA) to assess the biological turnover within each data set. Gradient lengths obtained for all organism groups (< 3.0 SD units) indicated the use of a linear model, therefore redundancy analysis (RDA) and partial RDA were carried out on each organism group. RDA, a direct gradient analysis, assumes that canonical ordination axes are linear combinations of environmental or pressure variables and provides an eigenvalue, an estimate of variance explained by the "species" and variables used in the analysis.

A forward selection, (cut off point of $P > 0.1$) was used to retain non-redundant subsets of environmental and pressure variables to explain taxonomic distribution patterns (Aguiar & Ferreira 2005). A Monte-Carlo permutation test (999 permutations) was run on the first axis eigenvalue and the "trace" (sum of all canonical eigenvalues) to test the significance of the environmental and pressure effects under analysis.

Partial Redundancy Analyses (Liu 1997) was used to determine decomposition of variance between environmental and pressure variables. Sets of forward selected environmental or pressure variables were used as covariables to discern the distribution of unique, shared and unexplained variance for each biological assemblage. All analyses were carried out with the software packages PRIMER (Version 6.1.8), STATISTICA (version 6), and CANOCO (version 4.5 for Windows).

Both Procrustean analysis (PROTEST software package - PROcrustean randomization Test: http://www.zoo.utoronto.ca/jackson/pro1.html) and the RELATE software package (PRIMER) were used to evaluate the degree of concordance between each biological assemblage and environmental and pressure data. Procrustean analyses superimpose, scale and rotate one data matrix upon the

Table 5.1 Environmental (E) and pressure (P) variables divided over three spatial scales (basin, reach and habitat) retained for subsequent statistical analyses for redundancy by comparing average Spearman Correlation Coefficients



Variable and spatial scale				Unit/expression	Data source
Basin					
E	1	Channel form	(CHAN_F)	1 = sinuate, 2 = constrained	GIS
E	2	Valley form	(VALFRM)	Class 0–4	GIS
P	1	Urban Area	(URB_A)	% Catchment area	GIS
P	2	Monocultures	(MONO_A)	% Catchment area	GIS
P	3	Agriculture	(AGRI_A)	% Catchment area	GIS
Reach					
E	3	Altitude	(ALT)	m.a.s.l	GIS
E	4	Number of bars	(BARS)	Count	RHS data
E	5	Average riparian width	(WDTRIP)	Metres	RHS data
E	6	Subchannels	(SUBCH)	Count	RHS data
E	7	Land use natural/semi-natural	(LU250_NAT)	Dummy variable	GIS 250 m bankside buffer
E	8	Land use scrub	(LU250_SCR)	Dummy variable	GIS 250 m bankside buffer
E	9	Average bank top height	(AVBKTP)	Metres	RHS data
P	1	Bank reinforced	(BK_RI)	Dummy variable	RHS data
P	2	Bank embanked	(BK_EM)	Dummy variable	RHS data
P	3	Land use agriculture	(LU250_AGR)	Dummy variable	GIS 250 m bankside buffer
P	4	Presence of Ford	(FORD)	Presence/absence	RHS data
P	5	Tipped debris	(TIP_D)	Presence/absence	RHS Data
P	6	Land use urban	(LU25 0_UR)	Dummy variable	GIS 250 m bankside buffer
P	7	Organic point discharge	(ORGP)	Count	GIS
Habitat					
E	10	pH	(PH)	Sorensen scale	Fish sampling site
E	11	Dissolved oxygen	(DO)	mg/l	Fish sampling site
E	12	Water velocity	(WVEL)	$m\ s^{-1}$	Macroinvertebrate sampling site
E	13	Conductivity	(COND)	µS/cm	Macroinvertebrate sampling site
E	14	Boulder/stone substrate	(BOLSTONB)	Visual estimate	Macroinvertebrate sampling site
E	15	Boulder/stone substrate	(BOLSTONF)	Visual estimate	Fish sampling site
E	16	Depth	(DEPTB)	Metres	Macroinvertebrate sampling site
E	17	Depth	(DEPTF)	Metres	Fish sampling site
E	18	Water temperature	(W_TEMPF)	° Celsius	Fish sampling site
E	19	Sand/silt substrate	(SASIC)	Visual estimate	Fish sampling site
E	20	Gravel substrate	(GRAV)	Visual estimate	Fish sampling site
P	8	Bank modification	(BNK_MOD)	Visual estimate	RHS/sample site
P	9	Channel modification	(CH_MOD)	Presence of features such as culverts, weirs or sluices	RHS/sample site
P	10	Banktop landuse agriculture	(BT_AG10)	Class	10 m bankside buffer
P	11	Banktop landuse pasture	(BT_RP10)	Class	10 m bankside buffer
P	12	Banktop landuse forestry	(BT_FR10)	Class	10 m bankside buffer

other until an optimal fit is found (Heino and others 2004; Jackson and Harvey 1993; Peres-Neto and Jackson 2001). Estimated residuals between original values and the derived best fit solution give the m^2 statistic. A low m^2 statistic indicates a good level of correspondence between data matrices (Paavola and others 2006). We used the PROTEST analysis package to assess the degree of concordance between the sample site scores of the first 3 axes of the PCAs run for hierarchically organized environmental and pressure matrices and the sample site scores derived from the first 3 axes of Canonical analyses (CA) run for each biological assemblage. The PROTEST permutation procedure

(999 permutations) was used to assess the statistical significance of the Procrustean fit between the two matrices (Paavola and others 2006; Peres-Neto and Jackson 2001).

The Mantel test, (the RELATE procedure in the PRIMER software package) was used to compare similarity matrices for each biological assemblage (Bray Curtis similarity) with similarity matrices of environmental and pressure data (Euclidean distance). Significance was assessed using a permutation procedure (999 permutations) applied to Spearman ranked transformed data. Peres-Neto and Jackson (2001) suggest that Procrustean analysis is more effective than the Mantel test for assessing concordance, since the former can be used on raw data or derived ordination solutions while the latter is based upon measures of distance or similarity.

5.4 Results

Environmental and Pressure Gradients

For clarity, only variables with loadings ≥ 0.5 on axes 1, 2 or 3 are described (Feld and Hering 2007). The environmental PCA had higher overall levels of explained variance compared to the pressure PCA, illustrated by the eigenvalues in particular of the first two axes (Table 5.2). The variables with the highest loadings along axis 1 of the environmental PCA were water velocity (WVELB), pH (PHF) and dissolved oxygen (DOF). The strong negative loadings of WVELB and PHF indicated the strong separation of sites upstream and downstream of the cofferdam. The highest, principally negative, loadings along axes 1 and 2 were water velocity (WVEL_B, -0.72 and -0.683, respectively) and pH (PH, -0.814 and 0.537, respectively). Habitats in the middle and upper reaches were characterised by higher water velocity levels while lower lying sites had higher pH, temperature and dissolved oxygen levels (DO_F). The higher downstream dissolved oxygen and pH levels can be partly attributed to the considerable daytime photosynthetic activity of dense stands of macrophytes and filamentous algae recorded at these sites. The water velocity and pH gradients were also evident along axis 2; however, the presence of reaches with natural land use (LU250_NA) characterised sites in the middle and upper sections of the Odelouca. Parameters with high loadings on axis 3 were related to habitat quality. Extremely long side bars (BARS; over 100m in length), probably associated with the reduced flow and resulting deposition of substrates (loading 0.779) tended to be associated with downstream sites, however smaller discrete side bars and mid channel bars were recorded at upstream sites as well. Sites with greater average riparian width (WDTRIP) tended to occur at sites upstream of the cofferdam. These sites also tended to be deeper (DEPTB, loading -0.574) with coarser boulder stone substrates (BOLSTONF, loading -0.504).

Table 2 - PCA eigenvalues and loadings (axes 1-3) for retained environmental and pressure variables divided over three spatial scales of habitat (H), reach (R) and basin.

		Axis 1	Axis 2	Axis 3
Environmental variables				
Eigenvalue		0.358	0.264	0.117
BARS	R	−0.385	−0.108	**0.779**
WDTRIP	R	0.103	−0.449	**−0.599**
LU250_NA	R	−0.041	**−0.506**	−0.311
WVELB	H	**−0.72**	**−0.683**	0.058
DEPTB	H	0.467	0.083	−0.574
pHF	H	**−0.814**	**0.537**	−0.199
DOF	H	**−0.512**	−0.032	−0.173
BOLSTONF	H	−0.236	−0.006	**−0.504**
Pressure variables				
Eigenvalue		0.271	0.182	0.117
BK_RI	H	**0.629**	−0.12	0.314
BK_EM	H	**0.64**	**0.582**	0.002
LU250_UR	R	0.288	−0.067	**0.767**
LU250_AG	R	**0.561**	−0.315	0.122
BNK_MOD	R	**0.79**	0.2	−0.454
BT_AG10	H	0.428	0.459	**0.575**
BT_RP10	H	**0.532**	**−0.73**	−0.071

The loadings of variables with values ≥ 0.5 are given in bold text. Acronyms are defined in Table 5.1

Several pressure PCA parameters revealed the strong gradient of physical disturbance of the river habitat and adjacent changes in land use at several spatial scales along axis one, namely bank reinforcement (BK_RI, loading 0.629) and embankment (BK_EM, loading 0.640), agricultural landuse (LU250_AG, loading 0.561), bank modification (BK_MOD, loading 0.790) and pasture (BT_RP10 loading 0.532). Sites suffering from these impacts were in the lower reaches of the Odelouca, mostly below the cofferdam. On axis 2, bank embankment (BK_EM, loading 0.582), and pastoral land use (BT_RP10, loading -0.730) further separated these sites. Axis 3 loadings were related to urbanisation and agricultural land use changes (LU250_UR, LU250_AG).

Biological assemblages

Organism abundance and diversity varied considerably among the different assemblages. Benthic macroinvertebrates were the most abundant organisms (total 40,346 individuals were sorted and identified) but the second most diverse group (34 species, 22 genera, 2 families, 1 sub family and one order were identified following taxonomic adjustment). The most abundant and frequently occurring order was the Diptera, in particular chironomids and simuliids.

Fish were the second most abundant (1,336 individuals) but least diverse assemblage (11 species were collected but only 8 species were subject to statistical analyses), reflecting the depauperate but endemic nature of Mediterranean fish assemblages. The most abundant species was the introduced alien, *Gambusia holbrooki* Girard. *Anguilla anguilla* L., was the most widely distributed fish species, occurring at 79% of the sample sites.

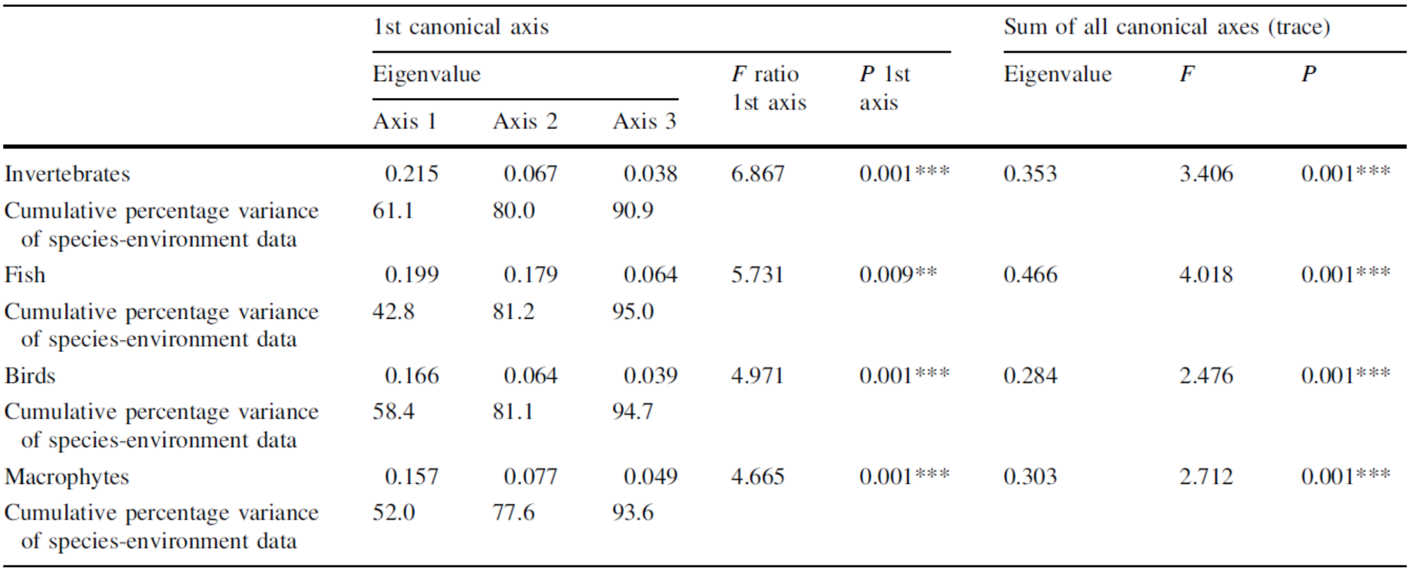
A total of 292 birds were surveyed represented by36 species (27 species were used in statistical analyses). The most frequent and abundant species was *Sylvia atricapilla* L, recorded at 90% of the sampling sites.

Plant biomass was harder to estimate due to the percentage cover estimate applied. However, a total of 71 species level taxa were described. Most frequently occurring species included *Mentha suaveolens* Ehrh *Oenanthe crocata* L, *Rubus ulmifolius* Scott (all three species were recorded 83.3% of sample sites) and *Salix salviifolia* Brot. (recorded at 90% of sampling sites)

RDA results

Extracted first axes and the trace for RDA ordinations were highly statistically significant for all assemblages (Monte Carlo test 999 permutations; Table 3.3). The highest levels of variance described by the axis 1 eigenvalues were for the invertebrate and bird biological assemblages. Strong environmental/pressure gradients were evident along axes 1 and 2 of the macroinvertebrate RDA (Figure 5.2a). The longitudinal pH (intra-set correlation r= -0.89) and altitude gradient (intra-set correlation r= 0.59) along axis 1 separated more disturbed downstream sites with higher pH values and macrophyte growth from less disturbed upstream sites. The axis two gradient separated faster flowing upstream sites (WVEL, intra-set correlation r= -0.45) from sites in agricultural areas (LU250_AGR intra-set correlation -0.87). More abundant taxa found in the lower reaches (higher pH axis 1), included *Baëtis* sp, Chironomidae *dugesia* sp, *Gyraulus* sp and *Simulium (Simulium)* sp. Taxa occurring more frequently at the other end of this gradient were *Capnioneura mitis* Despax tribe Tanytarsini, and *Chironomus (Chironomus) plumosus*-Gr. Taxa strongly associated with surrounding agricultural land use (axis 2) were *Stylaria lacustris* L (r= -0.82 p<0.05) and *Procloeon* sp (r= -0.58 p<0.05) while taxa associated with increased water velocity and altitude at the other end of axis 2 were *Dicronata (Dicronata)* sp, *Atherix* sp and *Onchychogomphus forcipatus*.

Table 5.3 Results of RDA and Monte Carlo permutations (F statistic, 999 permutations) for testing the significance of environmental and pressure variables to distribution patterns of benthic macroinvertebrates, fish, birds and macrophytes



	1st canonical axis					Sum of all canonical axes (trace)		
	Eigenvalue			*F* ratio 1st axis	*P* 1st axis	Eigenvalue	*F*	*P*
	Axis 1	Axis 2	Axis 3					
Invertebrates	0.215	0.067	0.038	6.867	0.001***	0.353	3.406	0.001***
Cumulative percentage variance of species-environment data	61.1	80.0	90.9					
Fish	0.199	0.179	0.064	5.731	0.009**	0.466	4.018	0.001***
Cumulative percentage variance of species-environment data	42.8	81.2	95.0					
Birds	0.166	0.064	0.039	4.971	0.001***	0.284	2.476	0.001***
Cumulative percentage variance of species-environment data	58.4	81.1	94.7					
Macrophytes	0.157	0.077	0.049	4.665	0.001***	0.303	2.712	0.001***
Cumulative percentage variance of species-environment data	52.0	77.6	93.6					

Axis 1 of the fish taxa ordination plot also described a similar pH gradient (Figure 5.2b; intraset correlation -0.828) but also a habitat complexity gradient (intraset correlation ALT= 0.63, BARS = 0.21). A shorter gradient related to channel modification and riparian gallery width was evident along axis 2 (intraset correlation CH MOD= -0.23, WDTRIP = -0.29). Threatened native species *S. aradensis* and *C. almacai* were clearly associated with upstream sites with greater in-channel habitat diversity and negatively associated with downstream sites suffering modification, including reduced flow and the physical barrier of the cofferdam. *Anguilla anguilla* L was more abundant at lower lying sites but widespread in the study area. Both alien (*Gambusia holbrooki* Girard and *Lepomis gibossus* L.) and native species (*Atherina boyeri* Risso, *Barbus sclateri* Günther and *Cobitis paludica* de Buen) were associated with channel modification and riparian gallery width. The presence of the native species *B. sclateri* with alien species reflects its preference for the pool-like conditions (Magalhães *and others* 2002), that predominate in the immediate area of the cofferdam. *G. holbrooki*, comprising just over 50% of the total catch, occurred exclusively at sites with reduced flow close to the cofferdam but where riparian galleries were still present.

Distinct bird species groups occurred in the ordination space (Figure 5.2c). A longitudinal gradient from higher lying to lower lying degraded sites was discernible (axis 1 intraset correlations pH r= 0.75, BK_RI r= 0.79; axis 2 intraset correlation ALT r=0.63, LU250_AG r=0.58). Species clearly associated with bankside disturbance (reinforcement) and reduced riparian cover at lower lying sites were *Ardea*

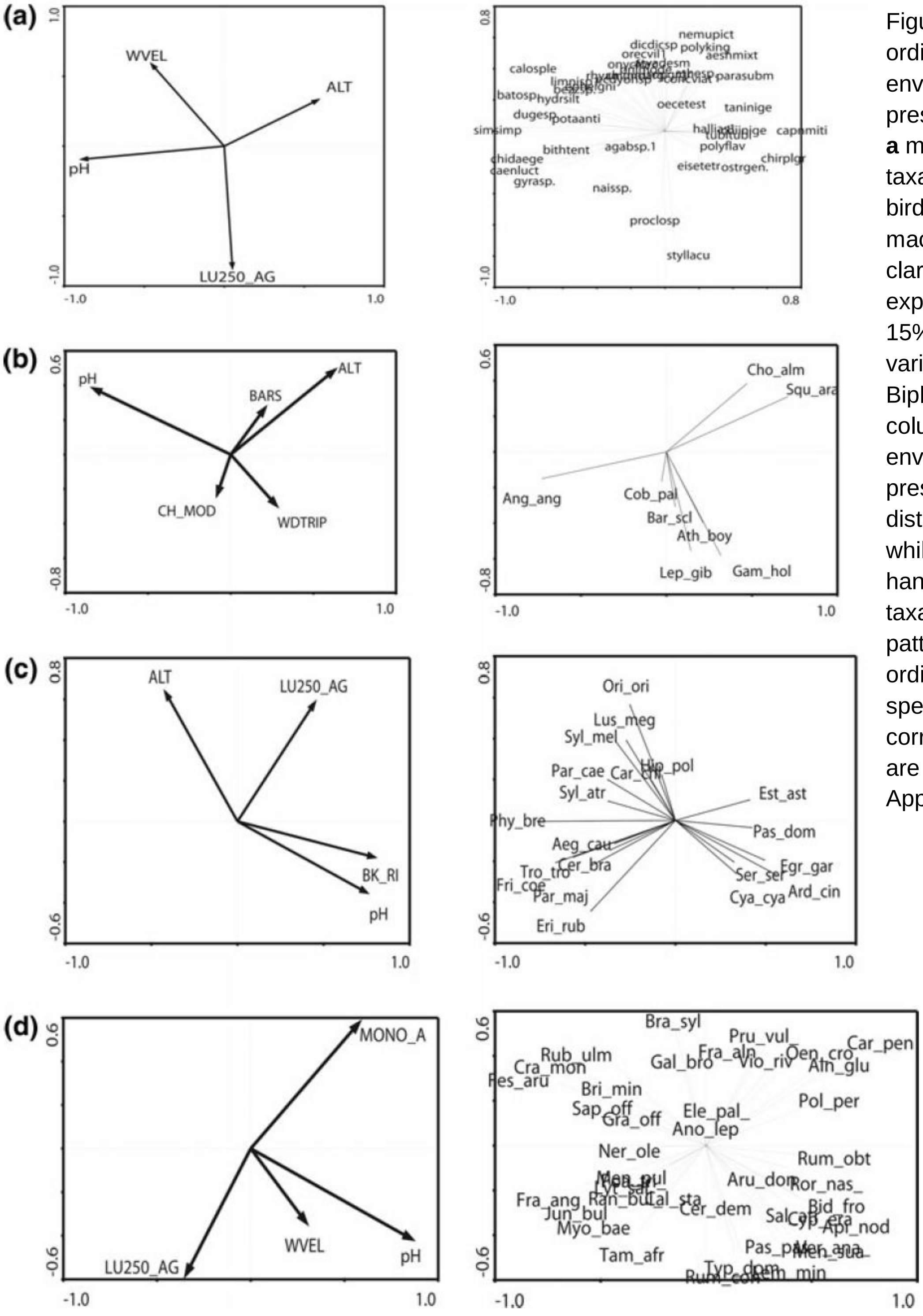


Figure 5.2 - RDA ordination of environmental and pressure variables on **a** macroinvertebrates taxa, **b** fish taxa, **c** bird taxa and **d** macrophyts. For clarity, only taxa explaining more than 15% explained variance are shown. Biplots in the left hand column describe environmental and pressure variable distribution patterns while biplots in the rgh handcolumn describe taxa distribution patterns in the ordination scale. Full species names and corresponding codes are gviven in Appendix III.

cinerea L, *Egretta garzetta* L and *Passer domesticus* L. Woodland species such as *Phylloscopus brehmii* von Homeyer, *Fringilla coelobs* L, *Parus major* L, and *Troglodytes troglodytes* L were negatively associated with areas affected by agriculture. A third distinct group of predominantly woodland/scrub species was also associated with less impacted upstream sites. Species significantly correlated with axis 2 of the plot were *Luscinia megarhynchos* Brehm; *Oriolus oriolus* L and *Sylvia melanocephala* Gmelin.

Distinct plant species distribution patterns were related to longitudinal hydric regime patterns and habitat degradation resulting from changes in land use (Figure 5.2d). The pH gradient (pH axis 1 intraset correlation r= 0.86, axis 2 intraset correlation r= -0.42), separated lower lying sites on the wider valley floor in agricultural areas from other main channel sites (LU250_AG axis 1 intraset correlation r= 0.34, axis 2 intraset correlation r= -0.58) and also narrow tributary sites with monocultures present in the area beyond the riparian zone (MONO_A axis 1 intraset correlation r= 0.58, axis 2 intraset correlation r= 0.58). Woody riparian species, tolerant of wetter conditions and occasional flooding, such as *Alnus glutinosa* (L.) Gaertner, *Fraxinus alnus* P. Mill, *Viola riviniana* (Rchb.) and *Prunella vulgaris* L. subspecies *vulgaris*, were associated with tributaries. Main channel sites affected by agriculture (bottom right hand side of the ordination biplot) were occupied by hardy woody shrubs tolerant of both aridity and disturbance such as *Tamarix africana* Poiret, *Fraxinus angustifolia* Vahl and *Juncus bulbosus* L. Degraded lower lying sites were characterised by the in-channel presence of *Apium nodiflorum* (L.), *Lemna minor* L., *Bidens frondosa* L. and *Cyperus eragrostis* Lam on the bankside. Species at less disturbed sites further upstream on the main channel included woody shrubs such as *Rubus ulmifolius* Scott, *Crataegus monogyna* Jacq and *Festuca arundinacea* Schreber.

Partial RDA

Distinct patterns of decomposition of variance by pRDA (Figure 5.3) occurred across the biological assemblages despite the universally high levels of unexplained variance levels (ranging from 53.4% for fish assemblages to 71.6% for birds) and very low levels of shared variance (environmental and pressure combined) for all groups (the highest value of just 3.9% attributable to bird assemblages). Levels of variance uniquely attributable to environmental parameters were consistently higher than those uniquely attributable to pressure variables for all biotic assemblages. This was particularly notable for fish assemblages (over 40% variance derived from the forward selected variables BARS, WDTRIP, PH and ALT, followed by the benthic macroinvertebrates (29.2%, WVEL, PH and ALT), macrophytes (17.7%, WVEL and PH) and birds (17.4%, variables PH and ALT). The pRDA forward selection results highlight the high level of redundancy between all assemblages for the parameters pH and altitude.

The highest levels of variance attributed uniquely to pressure variables were for bird (12.5%) and macrophyte (11.8%) assemblages. Groups of retained pressure variables for these two assemblages were associated with agricultural changes in land use and physical disturbance of the banks (birds). Levels of variance derived purely from single pressure variables for each aquatic faunal assemblage were markedly lower and related to agricultural changes in land use (macroinvertebrates 5.4%) and physical impacts on the river channel (presence of culverts, dams, weirs or sluices, fish 6.5%).

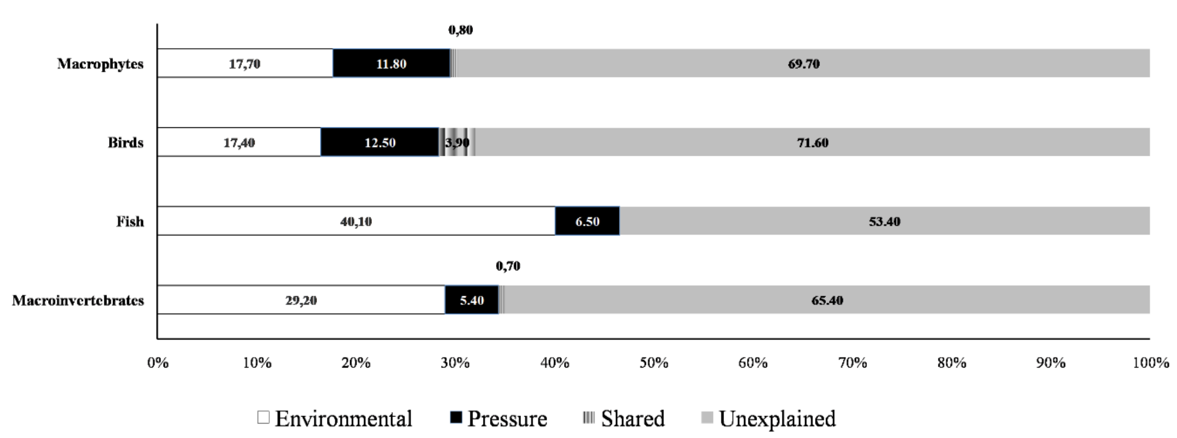
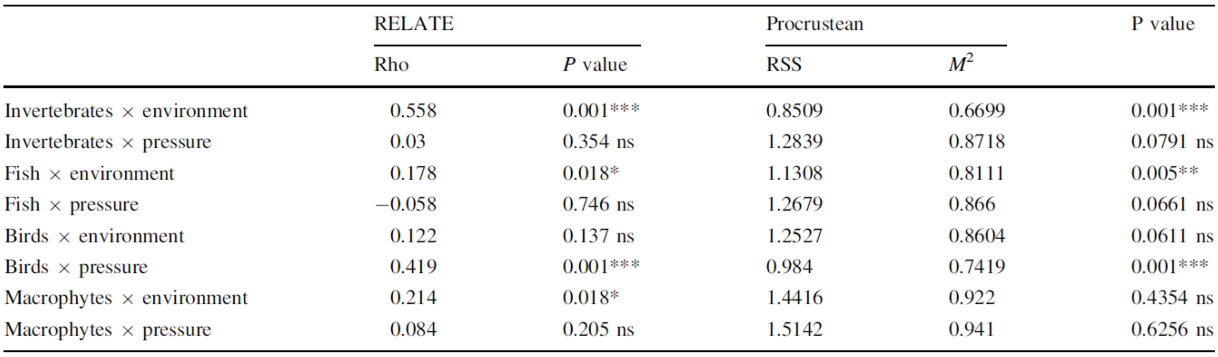


Figure 5.3 – Decomposition of variance across the biological assemblages in relation to forward selected environmental and pressure variables

Patterns of concordance for biological assemblages and environmental or pressure data were similar for both Procrustean & RELATE analyses (Table 5.4).

Highly significant levels of concordance occurred between macroinvertebrate assemblages and environmental parameters and bird assemblages and pressure parameters. Different levels of statistical significance of concordance occurred between fish assemblages and environmental

Table 5.4 – Results of concordance analyses between biological assemblages and environmental and pressure data using RELATE (Mantel Test) and Procrustean analyses.



	RELATE		Procrustean		P value
	Rho	*P* value	RSS	M^2	
Invertebrates × environment	0.558	0.001***	0.8509	0.6699	0.001***
Invertebrates × pressure	0.03	0.354 ns	1.2839	0.8718	0.0791 ns
Fish × environment	0.178	0.018*	1.1308	0.8111	0.005**
Fish × pressure	−0.058	0.746 ns	1.2679	0.866	0.0661 ns
Birds × environment	0.122	0.137 ns	1.2527	0.8604	0.0611 ns
Birds × pressure	0.419	0.001***	0.984	0.7419	0.001***
Macrophytes × environment	0.214	0.018*	1.4416	0.922	0.4354 ns
Macrophytes × pressure	0.084	0.205 ns	1.5142	0.941	0.6256 ns

The RELATE procedure compared similarity matrices while Procrustean analyses compared sample site values from the 1st three axes of Canonical Analyses (biological assemblages) and PCA (environmental and pressure variables). RSS = residual sum of squares

parameters, while weak levels of statistical significance were detected between macrophytes and environmental parameters by RELATE but not by Procrustean analyses. No statistically significant levels of concordance were detected between macroinvertebrates, fish and macrophyte assemblages and pressure variables or bird assemblages and environmental variables.

5.5 Discussion

The results of this study clearly highlights the importance of identifying environmental and pressure gradients acting upon river systems that will be subject to restoration, as well as the importance of *a priori* studies on biological assemblage response patterns and the implementation of typologically appropriate restoration measures.

PCA results showed that higher levels of variance were attributed to environmental parameters that described essentially longitudinal changes in the riverine environment. Benchmark sites in the mid section of the Odelouca, upstream of the cofferdam were characterised by mature riparian galleries and higher habitat diversity. Retained pressure variables indicated physical impacts to the riverbank such resectioning and reprofiling in areas dedicated to agriculture and, to a lesser extent, alterations to the channel itself. These impacts are in accordance with the observations of Hooke (2006) and Aguiar and Ferreira (2005) who state that principal human impacts in Mediterranean systems over the last century are mainly related to changes in land use and alterations in water and channel management. Hooke (2006) mentions the very long history of human impacts on Mediterranean systems. The fact that these systems are largely altered and have been so for such a long period of time may also explain the weak pressure gradient and poor association with biological assemblages that was detected in this study, i.e. most of the system has been disturbed with little variation for an extended period of (see Harding and others 2006).

However, the apparently strong environmental gradient is in fact a direct result of the innate covariance that exists between environmental and pressure variables (Allan 2004) that operate in European Mediterranean river systems (Díaz and others 2008; Hooke 2006). The longitudinal environmental gradient is intimately linked with basin orography and relief which in turn influence will patterns of human access and subsequent intervention. Large areas dedicated to agriculture occurred in the lower reaches of the Odelouca where the flood plain was considerably flatter and wider. Agricultural and urban areas also occurred at some sites upstream of the narrower, less accessible mid section where habitat disturbance was less pronounced and more natural conditions prevailed. The recorded instream habitat diversity is a direct result of the cyclic natural disturbance pattern of seasonal flood and drought typical of Mediterranean systems (Gasith and Resh 1999) to which many Mediterranean biological assemblages are adapted (Bonada and others 2007; Magalhães and others 2002; Díaz and others 2008), a fundamental concept of the Habitat Templet Theory (Southwood 1977; Southwood 1988) and river templet theory (Townsend and Hildrew 1994). Beche and others 2009 have shown that both invertebrate assemblage composition and native fish abundance levels are strongly affected by prolonged drought which influence flow in Mediterranean streams of California. These conditions also facilitated the establishment of an invasive fish species. The natural Mediterranean flow related disturbance pattern has been drastically altered at sites downstream of the cofferdam, which also lie in the wider areas of the catchment where levels of human intervention were found to be much higher. The cofferdam is also a physical barrier to fish migration. Thus, the effects of long term human intervention and the natural pattern of flow related disturbance characteristic of Mediterranean systems, result in complex gradients that are not as distinct as would initially appear. The “ideal” biological response to pressures should be type specific and distinct from responses to environmental gradients (Johnson and others 2006a). Our results show that this is not the case in selecting indicator species for typologically relevant restoration measures.

Multivariate RDA results indicated significant relationships between the species data and the explanatory variables for all biological assemblages and allowed the identification of key indicator species. However, pRDA results revealed low levels of explained variance across all assemblages, indicating that variables that could have better explained biological distribution patterns, in particular pressures, were not included in this study. Also, the separation of variables into groups of either environmental variables or pressure variables may also have affected results. For example, although classified as an environmental variable, pH is in fact a highly complex amalgam of changes in underlying natural biotic and abiotic gradients, changes in land use and enrichment (Hughes and others 2009, Townsend and others 1989). The longitudinal pH gradient detected in the Odelouca comprised a complex combination of background pH levels (catchment geology, biotic processes) and the result of changes in land use in the lower reaches due to human intervention. Other "environmental" variables such as the presence of bars (BARS) and riparian width (WDTRIP) can also be affected by changes in land use and water abstraction (i.e. very long side bars, formed by the reduced flow regime, were recorded downstream of the cofferdam). The lack of a measure of longitudinal connectivity, such as an estimate of distance from the cofferdam, may have improved the amount of variance described by pressures and the response of the biological assemblages, in particular the fish assemblage.

Results indicated that the variables we classified as "environmental" explained higher levels of variance than pressure variables across all groups, but that the associations were relatively weak, reflected in pRDA, RELATE and Procrustean results. The results of the aforementioned analyses were slightly stronger for macroinvertebrate and fish assemblages, giving information on important selected environmental descriptors of habitat quality that must be taken into account for typologically suitable restoration efforts, such as water velocity (macroinvertebrates), the presence of deposition bars and riparian galleries (fish). Further, Hughes and others (2008) found that the Habitat Quality Assessment Index (HQA - a measure of the structural diversity of natural features of wildlife interest along the river corridor, derived from RHS data), numbers of macroinvertebrates shredders and the EPT metric (Ephemeroptera Plecopter Trichoptera metric) were all significantly higher at benchmark sites, compared to sites below the cofferdam. Other studies on macroinvertebrate assemblages in Mediterranean systems clearly indicated the importance of flow patterns on macroinvertebrate traits and distribution patterns (Bêche and Resh 2007a; Bonada and others 2007). Microhabitat studies of *S. aradensis* and *I. almacai* revealed that each species occupied distinct microhabitats that varied with season and age (Santos and Ferreira 2008b). *I. almacai* preferred sheltered habitats with fine sediments in the autumn while younger nase occupied more exposed areas with coarser substrata than adult nase. *S. aradensis* preferred faster flowing habitats with coarser substrata in the spring compared to the rest of the year whereas younger chub tended to occupy slower flowing areas with finer sediments. These distribution patterns are intimately linked with seasonal flow patterns and associated processes of erosion and deposition.

The distinct distribution patterns across the fish assemblage highlighted the impact of the cofferdam on longitudinal connectivity, a factor that must be taken into account for effective restoration measures. *S. aradensis* and *I. almacai* were confined to the reaches of greater habitat quality, the eel *A. anguilla* was more abundant below the cofferdam and alien species occurred in high numbers at degraded sites affected by the structure of the cofferdam. These findings strongly contrast with surveys carried out in the Odelouca prior to cofferdam construction where *S. aradensis* and I*. almacai* were the most abundant and widespread species and the presence of alien species was negligible (Pires and others 2004). The predominance of alien species in the immediate vicinity of dams has also been recorded in “Mediterranean-type” river systems in California (Kondolf 1998) and in rivers suffering major flow alterations in the southwestern United States (Hughes and others 2005). Clearly the lack of a pressure variable quantifying disturbance in longitudinal connectivity affected results. According to Hooke (2006), connectivity within the different parts of Mediterranean river systems is a major driver of the extent and transmission of longitudinal changes.

Both the pRDA and RDA results clearly indicated a strong association between biological distribution patterns and the longitudinal pH gradient. Studies on environmental variables and invertebrates, fish and bryophyte assemblages in Finnish boreal streams (Paavola and others 2006) and physicochemical determinants of macroinvertebrate distribution in UK (Townsend and others 1989) streams have also identified the important role of pH. Studies tend to regard pH as a “natural” environmental variable when in fact it is an amalgam of both natural and anthropogenic factors. Background pH levels are attributable to catchment geology and biotic processes, but true pH levels are affected by multiple processes such as industrial airborne emissions, changes in land use and enrichment from agriculture and urbanization. This reveals not only the extremely complex nature of the longitudinal pH gradient in the lotic environment, but the inherent source of covariance contained within this parameter. Extremely dense stands of aquatic macrophytes and filamentous algae occurred at many of the lower lying sites in the Odelouca due to changes in land use (agriculture and associated bankside riparian clearance), reduced flow and increased sedimentation (reduced connectivity) and the natural longitudinal gradient (natural accretion, increased insolation due to the increased bankfull width). The higher pH levels recorded at these sites resulted from considerable photosynthetic activity as a result of the complex interplay of these factors.

Bird assemblages were most readily associated with the physical impacts related to land use changes. Similar responses have also been recorded in the bird assemblages on the Sado, a Mediterranean river system situated approximately 140 km north of the Odelouca (Moreira and others 1997). The results for the avifauna clearly highlight their increasingly recognized potential for assessing impacts across the wider river environment (Bryce and others 2002; Jansen and Robertson 2001b; Vaughn and others 2007). Explained variance attributed solely to pressure variables was highest in the bird assemblages, followed by the macrophytes. However, RELATE and Procrustean results were highly significant for the former assemblage but not at all significant for the latter. The results obtained for plant assemblages may be due again to the unintentional omission of descriptors of longitudinal connectivity that may have influenced distribution patterns.

The fundamental, multifunctional role of riparian vegetation in river systems is well known, ranging from flood prevention and sediment retention to seed recruitment, provision of shelter, habitat heterogeneity and allochthonous input. The maintenance of riparian vegetation is recognized as vital to the integrity of river ecosystems, although these ecotones are highly sensitive to environmental change (Gregory and others 1991, Naiman and Décamps 1997). Iberian riparian systems have been subject intensive agriculture and forestry, damming, abstraction and urban development, resulting in degraded habitat integrity, reduced longitudinal connectivity and isolated patches of riparian galleries clearly compromising wide scale lotic function (Aguiar and Ferreira 2005; Ferreira and others 2005). Riparian vegetation in semi arid areas tends to exhibit high levels of longitudinal variability in composition, species richness and density due to the environmental conditions (i.e. the strong annual and interannual patterns of flood and drought that characterise Mediterranean rivers) and land use in the immediate flood plain (Aguiar and Ferreira 2005; Raven and others 2009). The RDA and pRDA results reveal that the Odelouca is no different (retained environmental variables were W_VEL and pH, pressure variables were MONO_A and LU250_AG) and restoration efforts must take into account the current situation, namely the isolated but complex riparian stands in the mid section of the river, the influence of large scale environmental and pressure factors and the association of key biological assemblages with this ecotone.

The results of this study reveal a Mediterranean system clearly suffering from the effects of large scale impacts, namely reduced connectivity and habitat fragmentation due to changes in land use and damming. The Odelouca exhibits symptoms typical of river regulation. There is obvious reduced connectivity between upstream and downstream sites. Downstream sites are starved of sediment and variation in flow amplitude, leading to habitat fragmentation, impoverished habitat diversity, and disturbed lateral and vertical connectivity. System function and native biodiversity have been negatively affected and alien species have proliferated in areas of environmental degradation. These impacts are further compounded by changes in land use, resulting in physical alteration of the banks and channels and riparian clearance, further compromising lateral flood plain connectivity. Kondolf (1998) emphasized the importance of the study of both geomorphological and ecological processes during the whole restoration process. RHS and some SIG data provided an overview of geomorphologic processes, while field sampling of both aquatic and terrestrial assemblages prior to the restoration project provided us with information on patterns of changes in relation to environmental quality.

A decade ago Karr (1999) wrote that the "return of fish" was often listed as one of the principal biological goals of restoration. However, the restoration concept embraces a far broader and intricate picture that must take into account the spatial and temporal complexity of river systems and the multidisciplinarity approach necessary for the implementation of successful, typologically appropriate restoration measures. Ormerod (2004) fittingly states that "key species, the communities of which they are a part, and the ecological functions they provide must be the ultimate arbiters of restoration success". This study has shown, in the case of the Odelouca, that bird species can be used as indicators to detect bankside environmental degradation due to physical impacts and that distinct assemblages (comprising mainly woodland birds) occur at benchmark sites, where riparian integrity is good and instream habitat diversity is greater. Woody riparian communities are vital for increasing habitat diversity and providing refugia for faunal elements. Detailed phytosociological studies of these riparian communities should be made to contribute to appropriate planting initiatives during restoration. Selected benthic macroinvertebrate taxa and endemic fish species are important indicators of instream habitat quality, a direct consequence of less disturbed Mediterranean flow regime. All of these biological assemblages should be monitored during and after restoration and results can be presented in the form of metrics, *e.g.* number of macroinvertebrate shredder species, invertebrate EPT taxa, number of endemic fish species. Hughes and others (2009) found that metrics describing avian feeding or habitat preferences successfully separated bird species found in benchmark sites from degraded sites.

Palmer and others (2010) point out the importance of large scale factors for ensuring restoration success. The large scale impacts affecting the Odelouca require accordingly large scale but typologically relevant measures to restore connectivity and abiotic and biotic function once the dam is operational. Examples of measures include release of peak flow patterns consistent with Mediterranean flow patterns to re-establish connectivity, guarantee representative baseflow and re-establish typological patterns of erosion and deposition, thereby providing habitat diversity and refugia suitable for native fish species. Waterborne seed recruitment and propagules of riparian plant species (Dixon 2003), which will contribute to natural reestablishment of native riparian species, will also be intimately linked to restored flow patterns. The construction of a fish passage specifically designed to meet the needs of the species in question is fundamental to the success of restoration measures. Longitudinal connectivity is a major restoration goal, but lateral connectivity must also be considered to restore riparian function (Lake and others 2007). Eradication programmes of alien and invasive plant species such as *Arundo donax* L., which blights many degraded river banks across the Algarve region (Raven 2009), will be essential, followed by planting of native riparian species, grown from seed or propagated from the regional species pool. An equally essential part of the restoration process will be raising public and institutional awareness of the benefits of natural restoration measures on biodiversity.

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Chapter 6

GENERAL DISCUSSION

6. General Discussion

Twenty years have passed since the controversial article of Simberloff & Cox (1992) that highlighted the need for evidence on the efficacy of conservation corridors due to limited demonstration of the positive aspects of their usefulness and due to confirmation of negative effects of developing such an approach (e.g. dispersal of antagonistic species, diseases and invasive species). Though currently is established that conservation corridors are important conservation tools, most studies address the question imprecisely (Doerr *et al*., 2008). Despite encouragement for the development of ecological corridors and their restoration there is still a need to properly demonstrate their efficiency and the conditions and species for which they are useful.

Boulinier *et al.* (2001) found evidence of higher rates of local extinction and turnover in bird assemblages in more fragmented landscapes suggesting that communities function as meta-populations at regional scales, pointing out the importance of colonisations and re-colonisations from surrounding landscapes to local community dynamics. The habitat function of some corridors may be important to re-establish connectivity since some corridors may function as source to spill-over effects at landscape level (Damschen *et al.*, 2006; Brudvig *et al.*, 2009; Didham, 2010). Some habitats may not exhibit a conduit function, but the fact that they support more diverse and abundant communities at a landscape level may in fact lead them to contribute to maintain connectivity and decrease negative effects of habitat fragmentation (Bennet et al., 2014).

It has become widely assumed the important role and value that remnant riparian vegetation plays in maintaining assemblages of plants and animals in: agricultural landscapes, urban and suburban areas, tree plantations and productive forests (Bennet *et al*., 2014). The majority of these studies have been largely derived from site-based studies that compared richness, composition and/or abundance of biota at riparian forests with different features (*e.g*. width, management history, heterogeneity) (Bennet *et al*., 2014) but they only analysed good quality riparian vegetation (Saunders & Hobs, 1991; Whitaker & Montevechi, 1999; Jobin *et al.*, 2004; Palmer & Bennett, 2006; Cooke & Zack, 2009). Recently, Bennet *et al.* (2014) have analysed the benefits of riparian forest for bird assemblages at landscape level, yet only remnants of riparian forest surrounded by cleared farmland were studied.

6.1 Mediterranean riparian forests as habitat for birds

In Europe, riparian bird communities are essentially composed by non-riparian bird specialists (Roché, 2010) and only 4 species of riparian specialists (*Cinclus cinclus, Alcedo atthis, Motacilla cinerea* and *Histrionicus histrionicu*s) have been considered by Buckton & Ormerod (2002) global review. The information of European riparian bird communities is frequently sectorial, mostly limited only to waterbirds (*e.g*. Marchant & Hyde. 1980; Vaughan *et al*., 2007), non-easily accessible (especially from East European countries) (Roché, 2010) and scarce compared to USA and Australia for example. In European Mediterranean areas, despite some studies on riparian bird communities (Rabaça, 2004; Godinho *et al*., 2010), and the development of a bird index associated with river ecological quality (Larsen *et al.*, 2010) only one study compares matrix with riparian habitats (Pereira *et al*., 2014), moreover, no studies have been conducted to investigate if Mediterranean riparian areas support distinctive bird communities. Therefore, studies comparing riparian and adjacent non-riparian habitat are lacking and needed due to the several threats pending on these systems (*e.g*. land use pressures due to agriculture and livestock, climate change). It was expected that in xerofilous Mediterranean matrices riparian areas support more diverse and abundant species independently of their habitat quality.

In Chapter 2 “Living on the edge: the effect of habitat quality in breeding bird assemblages of riparian forests from SW Iberian Peninsula” and chapter 4 “The use of riparian strips as ecological corridors by songbirds: an experimental study prior and after clear-cut actions”, studies showed that riparian forests exhibited significantly richer and more abundant assemblages than their matrix pairs in both years for each study (2004 and 2006-chapter 2; 2007 and 2010, chapter 4). These results alone are well known and consistent with findings from other studies in other regions (Saunders & Hobs, 1991; Whitaker & Montevechi, 1999; Jobin *et al.*, 2004; Palmer & Bennett, 2006; Cooke & Zack, 2009; Bennet *et al*., 2014). However, results are also consistent independently of riparian habitat quality and matrix surrounding (chapter 2). Results are supported by four years of sampling with different observers, different surveys methods, so same conclusions are drawn in different years with slight species composition differences.

Despite the fact that riparian forests supported a richer and more abundant bird assemblage compared to other matrix typologies, results did not show that riparian forests in Mediterranean-type habitats supported a unique assemblage of birds (only two bird species unique of these habitats) in the breeding season. Results suggest and demonstrate that common European woodland birds (i.e. widespread species which are often abundant) use riparian areas as refuge habitats in xeric landscapes, thus are using them as habitat corridors, which is also supported by recent findings of Bennet *et al.* (2014) and Nimmo *et al.* (2015). In Australia, Bennet *et al.* (2014) have found that *riparian vegetation offers disproportionate benefits, relative to non-riparian vegetation, for conservation of woodland birds in highly modified agricultural landscapes*. Bearing in mind the general decline of European birds (Inger *et al*. 2015) and the fact that climate change is estimated to have impact on common resident birds (Sekercioglu *et al.,* 2008; Nimmo *et al*., 2016), our results support the idea that riparian areas are stable habitat corridors that play an important role at landscape level.

6.2 Impact of riparian habitat degradation

Habitat structure is a major determinant of bird community composition and many studies have demonstrated a link between riparian vegetation and birds in riparian zones, suggesting that changes in bird community composition can be predicted from changes in habitat structure (Rottenborn, 1999). Several studies demonstrated that fragmented riparian forests have conservation value for riparian bird species (Bentley & Caterall, 1997; Skagen *et al*., 1998; Palmer & Bennet, 2006; Seymour & Simmons, 2008; Bennet *et al*., 2014). Results from chapter 2 demonstrate that independently of riparian habitat quality, riparian forests still play an important role at landscape level as surrogate habitat for bird species. The comparison of matrix habitat types with different riparian habitat quality indicated that irrespective of the latter, the riparian forest plays a role in supporting bird communities at landscape level, as was mentioned by BouliIner *et al*. (2001) and Bennet *et al*. (2014).

Results revealed a negative tendency towards riparian degradation, with some riparian species showing a significant decrease when we compare good and very bad quality riparian forests. Godinho *et al*. (2010) and Larsen *et al*. (2010) have demonstrated the usefulness of bird assemblages as predictors of riparian degradation. Though chapter 3 “Ecological assessment of an intermittent Mediterranean river using multiple organism groups: the role of structure and function” results revealed significant levels of explained variance to species distribution patterns in relation to environmental and pressure variables for 4 biological assemblages (macrophytes, macroinvertebrates, fish and birds), in chapter 5

“Evaluating the Response of Biological Assemblages as Potential Indicators for Restoration Measures in an Intermittent Mediterranean River” analyses revealed that the avifauna was best associated with pressures acting upon the system rather than environmental characteristics. Chapter 5 results emphasize the importance of birds as indicators of wider river corridor health, marked structural and functional distribution patterns were linked with changes in land use (reflecting the influence of surrounding habitats), physical disturbance of the bankside and channel and habitat quality. The strong response of bird functional data to bank side/riparian habitat fragmentation has also been observed in another Mediterranean river systems (Moreira *et al*.,1997; Larsen *et al., 2010*) and in Australian river systems (Jansen & Robertson 2001). Our results further emphasise the importance of bird assemblages as effective indicators of the ecological integrity of the wider river landscape (Vaughn *et al.*, 2007).

Bock (1990) found that riparian stand size was relatively poor predictor of avian density or richness (both winter and summer) in Southeastern Arizona. Studies comparing riparian zones with substantially different mesic woody-vegetation structure have reported the greatest bird abundance and richness in habitats with the most complex vegetation structure (Sanders & Hobbs, 1991). Kilgo et al (1998) concluded that even narrow riparian zones can support an abundant and diverse avifauna, but Shirley & Smith (2005) observed that effects on riparian bird communities were greatest in very narrow contiguous forest buffers with high amounts of edge habitat, while Perry et al. (2011) found diverse responses among species to width of retained edge and structure of adjacent plantations. Clearly more detailed studies are needED to disentangle which characteristics of the riparian area can be used to enhance the use of riparian areas as corridors for birds. In fact, chapter 4 results suggest that poor quality corridors might function as ecological traps (Schlaepfer *et al*., 2002) at least for some species, since this is the first time that an impact on bird condition (weight and fat scores) due to riparian forest removal has been reported (though Green *et al*. (2011) have found two species which body mass declined as reservoir water levels increased, but direct influence of riparian quality in body condition could not be established). Burton *et al*. (2006) have reported impact on redshank (*Tringa totanus*) condition (weight decrease) due to habitat loss (dam filling – habitat un-availability).

Results from chapters 2 and 4 suggest that poor quality riparian forests play an important role at landscape level, and that these habitats are important surrogate habitat for birds. However, care should be taken when designing mitigation measures in riparian corridors since we should analyse the degrees of which structural characteristics of the riparian area continue to support fit animals. Care should be taken to analyse the extent of some characteristics of riparian forest (*e.g.* continuity and heterogeneity) that may not be able to provide birds the needed resources for example during migration (Finch & Young, 2000), when energy demands are greater and mortality rates are probably high (Green *et al*., 2011).

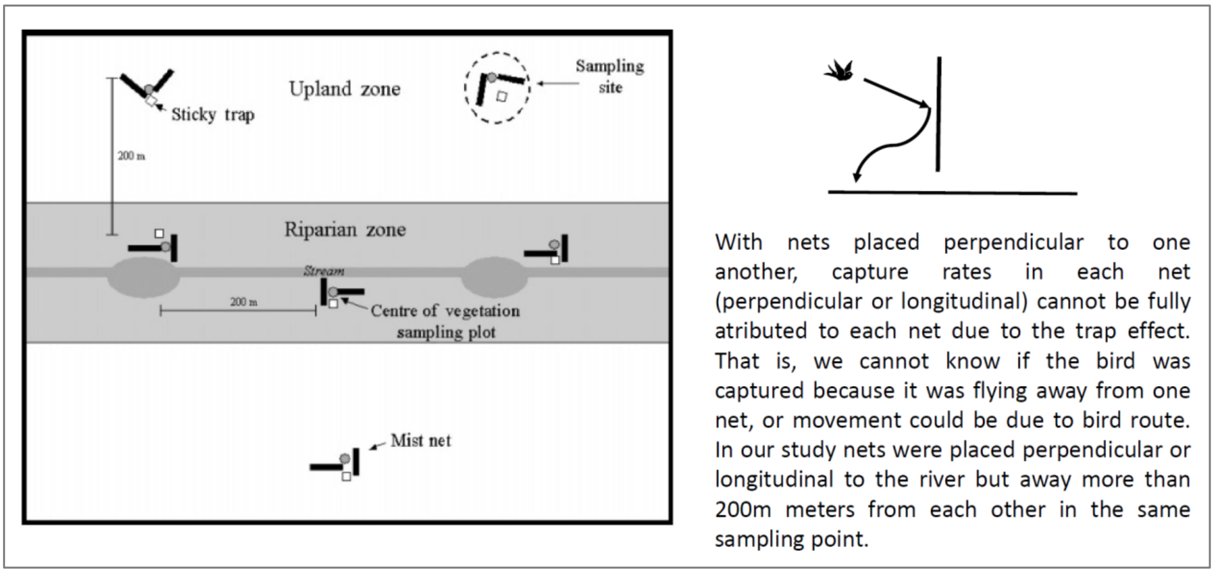
6.3 Mediterranean riparian forests as functional corridors

Riparian forests have long been considered ecological corridors for resident, breeding and migratory birds but few quantitative accounts in the literature support this idea and some even mention that corridors may not be important for vagile species like birds (Gilbert-Norton *et al*. 2010; Cushman *et al*., 2013). On the other hand, recent studies have demonstrated the importance of riparian areas as ecological corridors for specific bird species in the tropics (Gillies & St. Clair, 2008; Sekercioglu 2009) and for communities of American resident and migratory birds (Skagen *et al*., 1998; Means & Finch, 1999; Finch & Young, 2000; Skagen *et al*., 2005; Mosley *et al*., 2006).

We are aware that most studies that inferred corridor usage by comparing species diversity and abundance in connected and unconnected patches could not directly infer movement in corridors (Rosenberg *et al*., 1996; Doerr *et al*., 2010), because the presence of a species is not an indicator of movement use. Chapter 4 survey method allowed to determine birds' direction when captured in the riparian area and therefore it was possible to infer bird use of the riparian forest as ecological corridor (through nets placed perpendicular to the river) and as interface habitat (through nets placed parallel to the river). Mosley *et al*. (2006) made similar inferences although with a slightly different net scheme indicating that riparian areas may function as movement corridors for birds (Figure 6.1 and 4.2). Chapter 4 results reveal statistically significant evidence that riparian areas are used as movement corridors since lengthwise dimension captured higher bird rates than lateral dimension. Similar results were found by Mosley *et al.* (2006) in Canada during the fall migration period.

Considering that chapter 4 results indicate that more migratory birds were captured in this lengthwise dimension, it is likely that they use riparian areas as corridors because these habitats show lower resistance values to movement then the matrix. These findings are of great applicability to conservation since landscape resistance to movement is the first step to design biological corridors (Rosenberg *et al*., 1995; Cushman et al., 2013).

Following Rosenberg *et al*. (1995) definition, in chapter 4 is shown that *immigration to the target patch via corridor* (that is riparian habitat) *is greater than if the corridor were absent* (after clear-cut). The role of river corridors as stopover habitats for migratory birds on their route to African quarters was explored by Catry *et al*. (2009) for diurnal migrants crossing dry sectors of SW Iberia. Although they did not found a significant migratory corridor along the lower Guadiana River during fall migration, more studies are needed specially focusing on the role of riparian habitats as stopover sites.



With nets placed perpendicular to one another, capture rates in each net (perpendicular or longitudinal) cannot be fully atributed to each net due to the trap effect. That is, we cannot know if the bird was captured because it was flying away from one net, or movement could be due to bird route. In our study nets were placed perpendicular or longitudinal to the river but away more than 200m meters from each other in the same sampling point.

Figure 6.1 – Net scheme used by Mosley et al. (2006) compared to the net scheme used in this thesis (Figure 4.2).

6.4 Conservation implications

Though in chapters 2, 3 and 5 indicate that poor quality riparian forests still play an important role at landscape level, irrespective of habitat quality (supporting more diverse and abundant bird assemblages than the matrix), results in chapter 4 demonstrate that riparian forest removal greatly affects bird assemblages, indicating a general deterioration of suitable habitat for many bird species as a result of clear-cut. Moreover, it is important to take into account the possibility of negative, unintended consequences of bad quality riparian corridors. In the same way that corridors may facilitate movement of rare, endangered, or declining species, they may also increase mortality rates by not being able to provide sufficient energy for survival, particularly for migrant species (chapter 4). This is particularly important since migrant birds seem to have evolved special cognitive abilities that enable them to return to the same breeding, wintering and stopover sites in successive years (Mettke-Hofmann & Gwinner, 2003) and site-fidelity during migration has been reported in Iberian Peninsula for three species of warblers including Blackcap (Cantos & Tellería, 1994). So, it is important to conserve high-quality stopover sites, where birds can rapidly accumulate energy (Green *et al*., 2011). Skagen *et al*., (1998) advocate that all riparian patches (in southeastern Arizona) are important as stopover sites to migratory birds regardless of their size and degree of isolation. Chapter 4 results show that this may not stand for all species and all phenological categories in other parts of the world. Work is still needed to understand when and how riparian habitats may function as habitat corridors and which habitat features favor higher fitness of bird assemblages without causing negative effects, especially because impact on bird condition due to habitat changes has been detected elsewhere (Burton *et al*., 2006).

The results are also of importance at European level because common European bird species are declining (Inger *et al*., 2015) and SW Iberian Peninsula, is in the East Atlantic Flyway, which is an important area for trans-Saharan migrants known as the last stronghold before crossing the sea on their way to Africa (where major dam construction is foreseen Figure 6.2). Therefore, an articulation of Water Framework Directive, Habitats Directive and Birds Directive are imperative to achieve multipurpose goals and broader conservation strategies involving “out of countries perspective” and integrate all animals’ lifecycles habitats. Since rivers are the only natural linear continuous landscape units they may be used to contribute to the development of networks of ecological corridors.

Additionally, it is important to mention that riparian zones have acted as safe sites for regional flora during dry periods (*e.g*. Pleistocene droughts) (Naiman & Décamps, 1997), thus riparian forests restoration for multi-purposes can be important for promoting ecological connectivity in a changing climate (Krosby *et al*., 2010). Paleo-ecological evidence shows that response of organisms to past rapid climatic changes was migration to the environment space to which they are adapted (Huntley & Webb, 1989) and that riparian forests patches have acted as safe sites for regional flora during dry periods, as was demonstrated by Maeve & Kellman (1994) in Central America. Therefore, it seems reasonable to suggest that riparian forests can play an important role assisting biodiversity adaptation to climate change.

6.5 Final remarks

Irrespective of the corridor definition that we consider, that is habitat linked definition (Forman & Godron, 1986) or conduit definition (Simberloff *et al*., 1992), we were able to demonstrate that riparian songbird assemblages are different from the matrix, and that immigration to the target patch via riparian forest was greater in its presence than if the riparian forest was absent (Rosenberg *et al*., 1995 hypothesis). We support Hess & Fisher (2001) who suggested that conservationists and planners should consider all possible functions of corridors (habitat, conduit, filter, barrier, source, sink/trap) when designing them. More studies are needed on the structural characteristics that riparian forest should have to support viable and fitted assemblages of birds, as well as effects on birth, mortality, emigration and immigration. Focus is needed on demonstration of the positive aspects of linking nature reserves through riparian habitats, specially on the economic advantages of implementing broader conservation strategies (e.g. through scenario studies that include structural characteristics that demonstrate enhancement of biodiversity and its implementation costs) so that restoration objectives are met.

One of the most interesting results from the thesis is the demonstration that degradation of the riparian forest has impact on the bird condition, something that was pointed out in one article as possible yet no statistical difference was noted (Green et. al. 2011). A large number of dams are going to be built in regions where conditions are mesic (Figure 6.2), which reinforces results importance. Replicability, though difficult, will not be an issue and extrapolation to this context would be important for definition of compensations measures in riparian areas, migratory birds and the maintenance of alternative good quality riparian corridors at broader scales, that are clearly important for European trans-Saharan migrants.

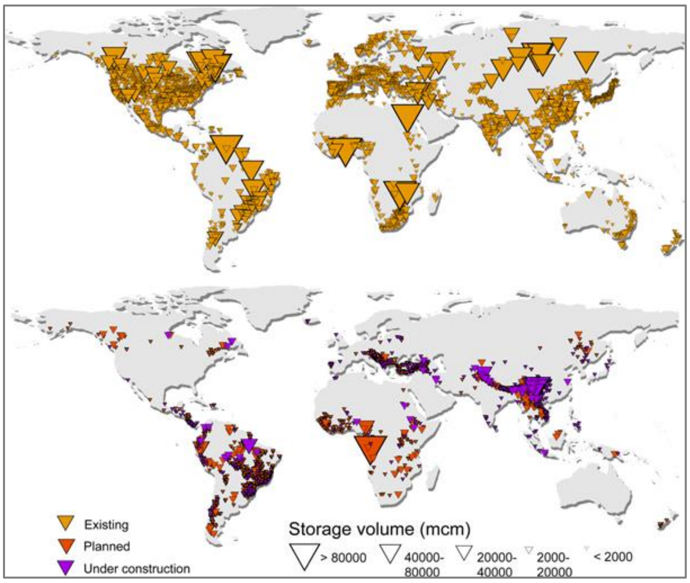


Figure 6.2 – Existing and future dams to be built (adapted from Gunther *et al.*, 2015).

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Appendices

Appendix I

Mean number of birds recorded (bird/m^2)*10^{-4} in 2004 and 2006 in the riparian habitat and in the matrix

Table 2.1 – Mean number of birds recorded (bird/m^2)*ha in 2004 and 2006 in the riparian habitat and in the matrix

	2004							2006						
Species	Riparian habitat			Matrix				Riparian			Matrix			
	Mean	SE	Sites	Mean	SE	Sites	*Total*	Mean	SE	Sites	Mean	SE	Sites	*Total*
Acr aru	0.828	0.462	**4**	0.003	0.003	**1**	*5*	0.214	0.161	**2**	_	_	**0**	*2*
Acr sci	0.548	0.325	**3**	_	_	**0**	*3*	0.371	0.242	**3**	0.005	0.003	**2**	*5*
Aeg cau	2.303	0.863	**9**	0.015	0.009	**4**	*13*	2.596	1.397	**6**	0.005	0.003	**2**	*8*
Alc ath	2.571	0.897	**12**	0.003	0.003	**1**	*13*	1.965	0.725	**9**	_	_	**0**	*9*
Ale ruf	_	_	**0**	0.010	0.006	**3**	*3*	_	_	**0**	0.017	0.007	**6**	*6*
Ana pla	2.199	1.387	**2**	0.005	0.004	**2**	*4*	_	_	**0**	0.025	0.025	**1**	*1*
But but	_	_	**0**	_	_	**0**	*0*	0.053	0.053	**1**	0.007	0.004	**3**	*4*
Apu apu	0.325	0.325	**1**	_	_	**0**	*1*	_	_	**0**	_	_	**0**	*0*
Ard cin	0.058	0.058	**1**	_	_	**0**	*1*	_	_	**0**	_	_	**0**	*0*
Asi otu	0.076	0.076	**1**	_	_	**0**	*1*	_	_	**0**	_	_	**0**	*0*
Bub ibi	_	_	**0**	0.040	0.020	**7**	*7*	_	_	**0**	_	_	**0**	*0*
But but	_	_	**0**	0.008	0.004	**3**	*3*	_	_	**0**	_	_	**0**	*0*
Car can	0.365	0.365	**1**	_	_	**0**	*1*	_	_	**0**	0.005	0.003	**2**	*2*
Car car	2.351	0.727	**12**	0.048	0.018	**12**	*24*	1.921	0.772	**12**	0.035	0.009	**13**	*25*
Car chl	3.401	0.905	**16**	0.033	0.012	**9**	*25*	8.919	2.693	**36**	0.074	0.012	**29**	*65*
Cer bra	_	_	**0**	0.020	0.008	**7**	*7*	0.770	0.394	**6**	0.071	0.031	**18**	*24*
Cet cet	14.395	2.210	**51**	0.025	0.010	**7**	*58*	10.644	1.614	**56**	0.005	0.005	**1**	*57*
Chl hib	_	_	**0**	_	_	**0**	*0*	_	_	**0**	0.025	0.025	**1**	*1*
Cha ale	0.895	0.895	**1**	_	_	**0**	*1*	_	_	**0**	_	_	**0**	*0*
Cha dub	_	_	**0**	0.003	0.003	**1**	*1*	_	_	**0**	_	_	**0**	*0*
Cic cic	_	_	**0**	0.015	0.007	**5**	*5*	0.025	0.023	**2**	0.373	0.370	**2**	*4*
Cir gal	_	_	**0**	0.003	0.003	**1**	*1*	_	_	**0**	_	_	**0**	*0*
Cis jun	4.819	1.523	**16**	0.143	0.020	**45**	*61*	_	_	**0**	0.213	0.132	**24**	*24*
Cla gla	_	_	**0**	_	_	**0**	*0*	0.138	0.119	**2**	_	_	**0**	*2*
Col pal	_	_	**0**	0.080	0.057	**2**	*2*	0.143	0.108	**2**	0.002	0.002	**1**	*3*
Cor cor	0.637	0.502	**2**	0.030	0.011	**8**	*10*	_	_	**0**	0.020	0.008	**7**	*7*
Cot cot	_	_	**0**	0.015	0.007	**5**	*5*	_	_	**0**	0.007	0.004	**3**	*3*
Cuc can	_	_	**0**	0.003	0.003	**1**	*1*	_	_	**0**	0.027	0.008	**11**	*11*
Cya cya	0.437	0.437	**1**	0.070	0.068	**2**	*2*	_	_	**0**	_	_	**0**	*0*
Del urb	0.576	0.471	**1**	0.003	0.003	**1**	*2*	_	_	**0**	0.002	0.002	**1**	*1*
Den maj	_	_	**0**	0.003	0.003	**1**	*1*	0.125	0.122	**2**	0.017	0.006	**7**	*9*
Egr gar	0.453	0.319	**2**	0.040	0.033	**3**	*5*	_	_	**0**	0.099	0.099	**1**	*1*
Emb cir	1.819	1.057	**3**	0.010	0.005	**4**	*7*	_	_	**0**	0.025	0.008	**10**	*10*
Eri rub	2.042	0.693	**9**	0.003	0.003	**1**	*10*	2.611	0.896	**14**	0.007	0.004	**3**	*17*
Est ast	16.520	5.469	**22**	0.023	0.012	**5**	*27*	5.415	4.258	**6**	0.007	0.004	**3**	*9*
Fal tin	0.178	0.178	**1**	_	_	**0**	*1*	_	_	**0**	_	_	**0**	*0*
Fri coe	2.987	1.240	**5**	0.078	0.015	**24**	*29*	2.438	0.864	**11**	0.284	0.132	**46**	*57*
Gal chl	0.172	0.172	**1**	0.005	0.005	**1**	*2*	0.606	0.304	**5**	_	_	**0**	*5*
Gal cri	_	_	**0**	0.025	0.010	**6**	*6*	_	_	**0**	_	_	**0**	*0*
Gal the	_	_	**0**	0.008	0.006	**2**	*2*	_	_	**0**	_	_	**0**	*0*
Galerida sp	_	_	**0**	0.003	0.003	**1**	*1*	_	_	**0**	0.017	0.006	**7**	*7*
Gar gla	0.182	0.182	**1**	_	_	**0**	*1*	0.209	0.163	**3**	_	_	**0**	*3*

Aqu fas	0.180	0.180	**1**	_	_	**0**	*1*	_	_	**0**	_	_	**0**	*0*
Him him	_	_	**0**	_	_	**0**	*0*	_	0.049	**0**	0.049	0.049	**1**	*1*
Hie pen	_	_	**0**	0.005	0.004	**2**	*2*	0.002	0.002	**1**	0.002	0.002	**1**	*2*
Hip pol	6.134	1.666	**20**	0.010	0.005	**4**	*24*	4.122	0.935	**24**	0.044	0.030	**7**	*31*
Hir dau	0.586	0.367	**3**	0.013	0.008	**3**	*6*	0.087	0.087	**1**	0.007	0.004	**3**	*4*
Hir rus	5.936	5.481	**4**	0.048	0.022	**8**	*12*	_	_	**0**	0.017	0.011	**4**	*4*
Lan mer	0.239	0.239	**1**	0.015	0.006	**6**	*7*	_	_	**0**	0.005	0.003	**2**	*2*
Lar fus	_	_	**0**	_	_	**0**	*0*	_	_	**0**	0.123	0.123	**1**	*1*
Lul arb	_	_	**0**	0.005	0.004	**2**	*2*	_	_	**0**	0.052	0.011	**20**	*20*
Lus meg	11.984	2.018	**44**	0.005	0.004	**2**	*46*	17.113	1.846	**83**	0.049	0.030	**8**	*91*
Mer api	0.190	0.190	**1**	0.023	0.010	**7**	*8*	0.509	0.304	**3**	0.098	0.051	**19**	*22*
Mil cal	0.355	0.234	**3**	0.098	0.022	**21**	*24*	2.197	0.701	**14**	0.335	0.141	**47**	*61*
Mot alb	2.757	1.948	**3**	0.003	0.003	**1**	*4*	_	_	**0**	_	_	**0**	*0*
Mil mig	_	_	**0**	_	_	**0**	*0*	_	0.002	**0**	0.002	0.002	**1**	*1*
Mot cin	_	_	**0**	_	_	**0**	*0*	0.034	0.034	**1**	0.005	0.003	**2**	*3*
Nic nic	_	_	**0**	_	_	**0**	*0*	_	_	**0**	0.002	0.002	**1**	*1*
Ori ori	0.398	0.398	**1**	0.003	0.003	**1**	*2*	_	_	**0**	0.002	0.002	**1**	*1*
Par cae	9.415	2.352	**25**	0.100	0.025	**23**	*48*	6.513	2.622	**21**	0.086	0.014	**32**	*53*
Par cri	0.180	0.180	**1**	_	_	**0**	*1*	_	_	**0**	0.002	0.002	**1**	*1*
Cya maj	4.099	1.374	**11**	0.043	0.013	**12**	*23*	0.652	0.289	**6**	0.015	0.007	**5**	*11*
Par sp	_	_	**0**	0.003	0.003	**1**	*1*	_	_	**0**	_	_	**0**	*0*
Pas dom	6.631	3.742	**8**	0.198	0.090	**15**	*23*	1.776	0.703	**10**	0.072	0.027	**19**	*29*
Pas mon	_	_	**0**	0.003	0.003	**1**	*1*	_	_	**0**	_	_	**0**	*0*
Pas sp	0.092	0.092	**1**	_	_	**0**	*1*	0.640	0.482	**2**	0.002	0.002	**1**	*3*
Pet pet	_	_	**0**	_	_	**0**	*1*	0.005	0.005	**1**	0.003	0.003	**2**	*2*
Pha col	_	_	**0**	0.003	0.003	**1**	*1*	_	_	**0**	_	_	**0**	*0*
Phy col	8.108	1.569	**30**	0.015	0.006	**6**	*36*	4.890	1.213	**27**	0.002	0.002	**1**	*28*

	2004							2006						
Species	Riparian habitat			Matrix				Riparian habitat			Matrix			
	Mean	SE	Sites	Mean	SE	Sites	*Total*	Mean	SE	Sites	Mean	SE	Sites	*Total*
Pic pic	_	_	**0**	0.008	0.006	**2**	*2*	_	_	**0**	_	_	**1**	*0*
Pic vir	0.750	0.750	**1**	_	_	**0**	*1*	_	_	**0**	0.002	0.002	**0**	*1*
Pla leu	_	_	**0**	_	_	**0**	*0*	_	_	**0**	0.074	0.074	**1**	*1*
Rapina sp	0.148	0.148	**1**	_	_	**0**	*1*	_	_	**0**	_	_	**0**	*0*
Ral aqu	_	_	**0**	_	_	**0**	*0*	0.171	0.171	**1**	_	_	**0**	*1*
Reg ign	_	_	**0**	_	_	**0**	*0*	0.290	0.177	**3**	_	_	**0**	*3*
Rip rip	_	_	**0**	0.010	0.008	**2**	*2*	0.102	0.102	**1**	0.007	0.007	**1**	*2*
Sax tor	4.333	2.060	**8**	0.088	0.015	**29**	*37*	1.788	0.773	**7**	0.067	0.012	**26**	*33*
Ser ser	7.420	1.881	**21**	0.093	0.021	**22**	*43*	4.800	0.856	**33**	0.194	0.052	**53**	*86*
Sit eur	0.076	0.076	**1**	0.025	0.008	**9**	*10*	_	_	**0**	0.037	0.010	**14**	*14*
Stu sp	0.075	0.075	**1**	0.023	0.010	**6**	*7*	_	_	**0**	0.002	0.002	**1**	*0*
Str dec	_	_	**0**	_	_	**0**	*0*	0.548	0.007	**3**	0.004	0.004	**3**	*6*
Str tur	_	_	**0**	_	_	**0**	*0*	0.717	0.015	**4**	0.006	0.006	**6**	*10*
Stu uni	_	_	**0**	0.065	0.058	**3**	*3*	1.304	0.537	**8**	0.042	0.010	**17**	*25*
Syl atr	7.447	1.735	**27**	0.030	0.021	**5**	*32*	8.626	1.430	**50**	0.002	0.002	**1**	*51*
Syl mel	9.372	2.161	**33**	0.088	0.018	**25**	*58*	3.238	1.148	**14**	0.105	0.051	**23**	*37*
Syl und	_	_	**0**	0.005	0.005	**1**	*1*	_	_	**0**	_	_	**0**	*0*
Sylvia sp	_	_	**0**	0.003	0.003	**1**	*1*	_	_	**0**	_	_	**0**	*0*

Tro tro	2.147	0.586	**16**	0.008	0.006	**2**	*18*	6.331	1.209	**44**	0.017	0.006	**7**	*51*
Tur mer	8.060	1.607	**33**	0.040	0.014	**11**	*44*	7.553	1.349	**46**	0.077	0.014	**27**	*73*
Tur vis	_	_	**0**	0.003	0.003	**1**	*1*	_	_	**0**	0.002	0.002	**1**	*1*
Upu epo	0.103	0.103	**1**	0.003	0.003	**1**	*2*	0.127	0.091	**2**	0.020	0.007	**8**	*10*
Sites rec			**52**			**64**	*78*	**44**	**46**	44			63	*69*
Sites unrec			**39**			**27**	*13*	**0**	**0**	47			28	*22*

Abreviations in Appendix IV

Appendix II

Results of ANOSIM pairwise comparisons between riparian habitat and different types of matrix

Table 2.3 – Results of ANOSIM pairwise comparisons between riparian habitat and different types of matrix. and results of SIMPER routine showing species contributing up to 70% to the several types of habitats surveyed and respective relative abundance of bird species.

		2004			2006		
Habitat	*R* stat	Species	Relative abundance (mean±SE)*10^{-4}	Cont (%)	Species	Relative abundance (mean±SE)*10^{-4}	Cont (%)
Riparian		Cet cet	2.6±0.2	22	Lus meg	3.3±1.8	36
2004- 116		Lus meg	2.6±0.2	19	Cet cet	2.1±1.6	16
2006- 142		Tur mer	1.4±0.1	10	Syl atr	1.7±1.4	11
		Syl mel	1.7±0.2	9	Tur mer	1.4±1.3	8
		Phy ibe	1.3±0.1	8			
		Syl atr	1.5±0.2	6			
***Eucaliptus* forest**		Syl atr	0.6±0.4	86	Ser ser	0.2±0.1	29
2004 -6	0.33n s				Par maj	0.2±0.1	29
2006- 5	0.586***				Mil cal	0.1±0.1	21
Montado forest		Sax tor	0.1±0.0	21	Ser ser	0.2±0.0	21
(*Quercus* sp)		Par cae	0.2±0.1	19	Fri coe	0.2±0.0	18
2004- 58	0.403***	Syl mel	0.1±0.0	14	Par cae	0.1±0.0	10
2006- 68	0.525***	Fri coe	0.1±0.0	10	Tur mer	0.1±0.0	7
		Ser ser	0.1±0.0	9	Mil cal	0.1±0.0	7
					Syl mel	0.1±0.0	6
Pasture		Mil cal	0.3±0.1	45	Mil cal	0.2±0.1	37
2004- 22	0.434***	Cis jun	0.2±0.1	38	Cis jun	0.1±0.1	21
2006- 16	0.556***				Ser ser	0.2±0.1	13
Olive groves		Pas dom	0.6±0.3	69	Mil cal	0.4±0.1	7
2004- 22	0.475***	Mil cal	0.2±0.1	12	Pas dom	0.1±0.1	4
2006- 8	0.584***				Tur mer		4
Rice fields		Cis jun	0.5±0.2	65	Cis jun		33
2004- 7	0.475***	Pas dom	0.8±0.5	19	Sax tor	0.1±0.1	28
2006- 6	0.55***				Fri coe	0.2±0.1	12
Sand extraction		Cis jun	0.3±0.1	100	Mil cal	0.3±0.1	75
2004- 3	0.469***				Cis jun	0.3±0.9	25
2006- 4	0.49ns						
Irrigated crops		Col pal	2.5±1.5	71	Mil cal	1.5±1.5	50
2004- 4	0.395*	Cis jun	0.2±0.1	29	Syl mel	0.8±1.6	50
2006- 4	0.385ns	-	-	- -	-	-	-
Pine		Cis jun	0.6±0.1	100	Par cae	0.4±0.1	35
2004- 5	0.494***				Lul arb	0.3±0	34
2006- 3	0.491***				Ser ser		11

Appendix III

(a) Final list of metrics calculated for each organism and

(b) List of taxa recorded for each organism group

Appendix III. (a) Final list of metrics calculated for each organism group and retained following Spearman correlation analysis and PCA. (b) List of taxa recorded for each organism group with individual taxon codes and distribution data (% of total abundance) in the upper reaches. lower reaches and tributaries of the Odelouca River (● 1-35%; ●36-60% ● >61%).

(a)

Organism group and metric type		**Code**	**Data transformation**
Macroinvertebrates			
Current preference	% Rheophilous taxa	%_RHEOPHIL	Arcsin√
Microhabitat preference	% Lithal taxa	% LITHAL	Arcsin√
Feeding preference	% Filter feeders	%_FILT_FEED	Arcsin√
Taxonomic group	Number of Trichoptera taxa	N_TRICH	log (x+1)
Taxonomic group	Number of EPT Taxa	EPT	log (x+1)
Taxonomic group	Number of EPT/Diptera	EPT/DIPT	log (x+1)
Fish			
Taxonomic group	% exotic individuals	% EXO_IND	Arcsin√
Taxonomic group	% native individuals	% NAT_IND	Arcsin√
Feeding preference	%_insectivore_individuals	%_INS_IND	Arcsin√
Habitat preference	number_eurytopic_individuals	N_EURY_IND	log (x+1)
Feeding preference	number_piscivore_species	N_PISC_SP	log (x+1)
Birds			

(a)

Organism group and metric type		**Code**	**Data transformation**
Diversity	Simpson	SIMPSON	log (x+1)
Feeding preference	Number seed eating species	N_SEED_SP	log (x+1)
Feeding preference	% seed eating individuals	%_SEED_IND	Arcsin√
Feeding preference	Number herbivore individuals	N_HERB_IND	log (x+1)
Feeding preference	% piscivore individuals	%_PISC_IND	Arcsin√
Habitat preference	Number tree dwelling individuals	N_TREE_IND	log (x+1)
Habitat preference	% open ground individuals	%_GRND_IND	Arcsin√
Macrophytes			
Habitat	Number pteridophyte species	N_PTER_SP	log (x+1)
Taxonomic group	% Coverage by exotic species	%_EXO_SP	Arcsin√
Taxonomic group	Number woody species	N_WOOD_SP	log (x+1)
Enrichment	Mean Trophic Rank	MTR	log (x+1)

(b)

		upper reaches	lower reaches	tributaries
Macroinvertebrates				
Aeshna mixta Latreille	aeshmixt	•		●

(b)

		upper reaches	lower reaches	tributaries
Agabus sp.	agabsp.1	●	•	•
Ancylus fluviatilis Müller	ancyfluv	•	●	•
Atherix sp.	athesp.	●		●
Atyaephyra desmarestii Millet	atyadesm	●	•	•
Baëtis sp.	batosp.	•	●	●
Bezzia sp.	bezzsp.		●	●
Bithynia (Bithynia) tentaculata L.	bithtent		●	
Branchiura sowerbyi Beddard	bransowe	●		
Brillia bifida Kieffer	brilmode	•		●
Caenis luctuosa Burmeister	caenluct	•	●	•
Calopteryx splendens Harris	calosple		•	●
Capnioneura mitis Despax	capnmiti	●	•	•

(b)

		upper reaches	lower reaches	tributaries
Centroptilum luteolum Müller	centlute		⬤	
Chimarra marginata L.	chimmarg	⬤	•	•
Chironomidae	chidaege	•	⬤	•
chironomini	chiinige	•	•	●
Chironomus (Chironomus) plumosus-Gr.	chirplgr	•	⬤	•
Cloëon dipterum L.	cloedipt	⬤	•	
Cloëon simile Eaton	cloesimi	•	⬤	
Conchapelopia viator Kieffer	concviat	●	•	•
Dicronata (Dicronata) sp	dicdicsp	•		⬤
Dugesia sp	dugesp.	•	⬤	•
Ecdyonurus sp	ecdyonsp	•	•	⬤
Eiseniella tetraedra Savigny	eisetetr	⬤	•	•

(b)

		upper reaches	lower reaches	tributaries
Empididae	empidige	●	•	●
Glossiphonia sp	glossisp	⬤	●	
Gyraulus sp	gyrasp.	•	⬤	
Habrophlebia fusca Curtis	habrfusc	●	•	⬤
Haliplus (Liaphlus) sp	hallias1	⬤	•	
Hydroporus sp1	hydrosp1	●		●
Hydropsyche angustipennis (Curtis)	hydrangu	●	●	•
Hydropsyche siltalai Doehler	hydrsilt	•	⬤	•
Hydropsyche tibialis McLachlan	hydrtibi	⬤		
Hydroptila sp	hytilasp	⬤	•	•
Isoperla grammatica (Poda)	isopgrgr	⬤	•	•
Leuctra geniculata Stephens	leucgeni		•	⬤

(b)

		upper reaches	lower reaches	tributaries
Limnius sp	limnisp1		●	⬤
Limnophila sp	libnotsp			⬤
Lymnaea stagnalis L.	lymnstag			⬤
Lype reducta Hagen	lyperedu			⬤
Micropsectra sp	mictrasp			⬤
Nais sp	naissp.	●	●	•
Nemurella pictetii Klapálek	nemupict	⬤		
Oecetis testacea (Curtis)	oecetest	⬤	•	
Onychogomphus forcipatus L.	onycforc	•	•	⬤
Orectichilus vilosus (Müller)	orecvil1	●	•	•
Ostracoda	ostrgen.	⬤		
Oulimnius sp	Oulisp 1		•	⬤

(b)

		upper reaches	lower reaches	tributaries
Paraleptophlebia submarginata (Stephens)	parasubm	•	•	⬤
Physa fontinalis (L.)	physfont	•	⬤	
Physella (Costatella) acuta (Draparnaud)	physache	●		●
Pisidium sp	casesp.		⬤	•
Polycentropus flavommaculatus (Pictet)	polyflav	⬤	•	
Polycentropus kingi McLachlan	polyking	⬤		
Polycentropus sp	poopussp	•		⬤
Polypedilum (Polypedilum) sp	poposp.	⬤		
Potamopyrgus antipodarum Gray	potaanti	•	•	⬤
Procambarus clarkii (Girard)	procclar	●	⬤	
Procloeon sp	proclosp	●	●	
Rhyacophila munda McLachlan	rhyamund	•	•	⬤

(b)

		upper reaches	lower reaches	tributaries
Serratella ignita (Poda)	epheigni		•	⬤
Simulium (Simulium) sp	simsimp	•	⬤	•
Stenelmis sp	stensp.1		⬤	
Stylaria lacustris L.	styllacu	●	●	
Tabanus sp	tabasp.	•		⬤
Tanypodinae	tannaege	●		●
Tanytarsini sp	taninige	⬤	•	•
Thraulus bellus Eaton	thrabell	⬤		
Tinodes sp	tinosp.	•		⬤
Tipula (Tipula) sp	tipusp.	⬤	•	•
Tubifex tubifex Müller	tubitubi	●	•	●
Fish				
Anguilla anguilla L.	ang_ang	•	⬤	•

(b)

		upper reaches	lower reaches	tributaries
Atherina boyeri Risso	Ath_boy	●		
Barbus sclateri Günther	Bar_scl	•	•	●
Iberochondrostoma almacai (Coelho. Mesquita & Collares-Pereira)	Cho_alm	●		
Pseudochondrostoma polylepis (Steindachner)	Cho_pol		●	
Cobitis paludica (de Buen)	Cob_pal	●	•	•
Gambusia holbrooki Girard	Gam_hol	●	●	
Lepomis gibbosus L.	Lep_gib	•	●	
Micropterus salmoides (Lacepéde)	Mic_sal		●	
Mugil spp	Mug_spp		●	
Squalius aradensis (Coelho. Bogutskaya. Rodrigues & Collares-Pereira)	Squ_ara	●		•
Birds				
Aegithalos caudatus L.	Aeg_cau	●		
Alcedo atthis L.	Alc_att	•	●	

(b)

		upper reaches	lower reaches	tributaries
Anas platyrhynchos L.	Ana_pla		⬤	
Ardea cinerea L.	Ard_cin		⬤	
Carduelis cannabina L.	Car_can	⬤		
Carduelis carduelis L.	Car_car	⬤	•	
Carduelis chloris L.	Car_chl	⬤	•	
Certhia brachydactyla Brehm	Cer_bra	⬤	•	•
Cettia cetti Temminck	Cet_cet	●	●	•
Charadrius dubius Scopoli	Cha_dub	●	●	
Cisticola juncidis Rafinesque	Cis_jun		⬤	
Cyanopica cooki Pallas	Cya_cya	•	⬤	
Egretta garzetta L.	Egr_gar		⬤	
Erithacus rubecula L.	Eri_rub	●	•	•
Estrilda astrild L.	Est_ast	●	●	

(b)

		upper reaches	lower reaches	tributaries
Fringilla coelebs L.	Fri_coe	⬤	•	•
Gallinula chloropus L.	Gal_chl		⬤	
Garrulus glandarius L.	Gar_gla	•	●	•
Hippolais polyglotta Vieillot	Hip_pol	●	●	•
Lullula arborea L.	Lul_arb			⬤
Luscinia megarhynchos Brehm	Lus_meg	⬤	•	•
Miliaria calandra L.	Mil_ca	•	●	●
Motacilla alba L.	Mot_alb		⬤	
Motacilla cinerea Tunstall	Mot_cin	•	●	●
Oriolus oriolus L.	Ori_ori	⬤	•	
Cyanistes caeruleus L.	Par_cae	⬤	•	•
Parus major L.	Par_maj	●	•	•
Passer domesticus L.	Pas_dom	•	⬤	

(b)

		upper reaches	lower reaches	tributaries
Phylloscopus ibericus (Ticehurst)	Phy_bre	⬤	•	•
Regulus ignicapillus Temminck	Reg_ign	•		⬤
Saxicola torquata L.	Sax_tor		⬤	
Serinus serinus L.	Ser_ser	•	⬤	
Sylvia atricapilla L.	Syl_atr	●	•	•
Sylvia melanocephala Gmelin	Syl_mel	⬤	•	•
Troglodytes troglodytes L.	Tro_tro	●	•	●
Turdus merula L.	Tur_mer	●	●	•
Macrophytes				
Ageratina adenophora (Sprengel) R. King & H. Robinson	Age_ade		•	⬤
Alnus glutinosa (L.) Gaertner	Aln_glu	•	•	●
Anogramma lepthophylla (L.) Link	Ano_lep	•	⬤	•
Apium nodiflorum (L.) Lag.	Api_nod		⬤	•

(b)

		upper reaches	lower reaches	tributaries
Arundo donax L.	Aru_don	●	•	•
Asplenium onopteris L.	Asp_ono	•		⬤
Bidens frondosa L.	Bid_fro		⬤	•
Brachypodium slyvaticum (Hudson) Beauv.	Bra_syl	●	•	●
Briza minor L.	Bri_min	⬤	•	
Bryophyte	Briofito	⬤	•	•
Callitriche stagnalis Scop.	Cal_sta	●	●	•
Carex pendula Hudson	Car_pen			⬤
Ceratophyllum demersum L.	Cer_dem	●	●	•
Corrigiola litoralis L.	Cor_lit	●	●	
Crataegus monogyna Jacq.	Cra_mon	⬤	•	
Cynodon dactylon (L.) Pers.	Cyn_dac	⬤	•	•
Cyperus eragrostis Lam.	Cyp_era	•	●	•

(b)

		upper reaches	lower reaches	tributaries
Cyperus longus L.	Cyp_lon	●	●	•
Cyperus rotundus L.	Cyp_rot	●	●	
Dactylis glomerata L. subsp. *hispanica* (Roth.) Nyman	Dac_glo_his	⬤	•	•
Eleocharis palustris (L.) Roemer & Schultes subsp. *Palustris*	Ele_pal_pal	⬤	•	
Erica lusitanica Rudolphi	Eri_lus	⬤	•	•
Festuca arundinacea Schreber	Fes_aru	⬤	•	
Frangula alnus Miller	Fra_aln	●		●
Fraxinus angustifolia Vahl.	Fra_ang	⬤	•	
Galium broteroanum Boiss & Reuter	Gal_bro	●		●
Galium palustre L. subsp. *tetraploideum* Clapham	Gal_pal_tet	●	●	
Gratiola officinalis L.	Gra_off	⬤		•
Holcus lanatus L.	Hol_lan	●	•	•
Isolepis setacea (L.) R. Br	Iso_set	⬤	•	

(b)

		upper reaches	lower reaches	tributaries
Juncus bufonius L.	Jun_buf	⬤		•
Juncus bulbosus L.	Jun_bul	⬤		
Juncus effusus L.	Jun_eff		●	●
Lemna minor L.	Lem_min		⬤	•
Lotus pedunculatus Cav.	Lot_ped	⬤	•	•
Ludwidgia palustris (L.) Elliot	Lud_pal	⬤	•	
Lythrum junceum Banks & Sol.	Lyt_jun	⬤	●	
Lythrum salicaria L.	Lyt_sal	⬤	•	•
Mentha aquatica L.	Men_aqu	•	⬤	
Mentha pulegium L.	Men_pul	●	●	
Mentha suaveolens Ehrh.	Men_sua	●	●	•
Myosotis baetica (Pérez Lara) Rocha Afonso	Myo_bae	⬤	•	•

(b)

		upper reaches	lower reaches	tributaries
Nerium oleander L.	Ner_ole	⬤	•	•
Oenanthe crocata L.	Oen_cro	●	•	●
Paspalum paspalodes (Michx) Scribner	Pas_pas	•	⬤	•
Poa trivialis L. subsp. *trivialis*	Poa_tri_tri	⬤	•	•
Polypogonum monspeliensis (L.) Desf.	Pol_mon	●	●	
Polygonum persicaria L.	Pol_per	•	●	●
Prunella vulgaris L. subsp. *vulgaris*	Pru_vul_vul	●		●
Pseudognaphalium luteo-album (L.) Hilliard & B. L. Burtt	Pse_lut_alb	•	⬤	
Pulicaria uliginosa Link	Pul_uli	⬤	•	•
Ranunculus bulbosus L. subsp. *aleae* (Willk.) Rouy & Foucaud	Ran_bul_ale	⬤	•	
Ranunculus peltatus Schrank subsp. *baudotii (*Godron) C. D. K. Cook	Ran_pel_bau	●	●	•
*Rorippa nasturtium-aquatica (*L.) Hayek	Ror_nas_aqu		⬤	•
Rubus ulmifolius Scott	Rub_ulm	⬤	•	•

(b)

		upper reaches	lower reaches	tributaries
Rumex conglomeratus Murray	Rum_con	●	●	•
Rumex obtusifolius L.	Rum_obt		●	●
Salix atrocinerea Brot.	Sal_atr		⬤	
Salix salviifolia Brot.	Sal_sal	●	•	•
Samolus valerandi L.	Sam_val	●	●	
Saponaria officinalis L.	Sap_off	⬤	•	•
Schoenoplectus lacustris (L.) Palla	Sch_lac	⬤	•	
Scirpoides holoschoenus (L.) Sojak	Sci_hol	●	●	•
Scrophularia auriculata L.	Scr_aur	•	•	•
Scrophularia canina L.	Scr_can	⬤	•	
Selaginella denticulata (L.) Spring	Sel_den	⬤		•
Solenopsis laurentia (L.) C. Presl.	Sol_lau	⬤	•	
Tamarix africana Poiret	Tam_afr	●	●	•

(b)

		upper reaches	lower reaches	tributaries
Tamus communis L.	Tam_com	•	•	●
Typha domingensis (Pers.) Steudel	Typ_dom	•	●	
Veronica anagallis-aquatica L.	Ver_ana_aqu	•	●	•
Viola riviniana Reichenb.	Vio_riv	•		●

Appendix IV

List of abreviations

Species	**Acronim**
Aegithalos caudatus	Aeg cau
Actytis hypoleucos	Act hyp
Alcedo atthis	Alc att
Alectoris rufa	Ale ruf
Anas platyrhynchos	Ana pla
Apus apus	Apu apu
Apus melba	Apu mel
Ardea cinerea	Ard cin
Athene noctua	Ath noc
Bubulcus ibis	Bub ibi
Buteo buteo	But but
Carduelis cannabina	Car can
Carduelis carduelis	Car car
Carduelis chloris	Car chl
Certhia brachydactyla	Cer bra
Cettia cetti	Cet cet
Charadrius dubius	Cha dub
Circaetus gallicus	Cir gal
Cisticola juncidis	Cis jun
Columba palumbus	Col pal
Columba livia	Col liv
Coturnix coturnix	Cot cot
Cyanopica cooki	Cya cya
Delichon urbicum	Del urb
Dendrocopos major	Den maj
Dendrocopos minor	Den min
Egretta garzetta	Egr gar

Emberiza cia	Emb cia
Emberiza cirlus	Emb cir
Erithacus rubecula	Eri rub
Estrilda astrild	Est ast
Falco tinnuculus	Fal tin
Fringilla coelebs	Fri coe
Galerida sp.	Galerida sp
Gallinula chloropus	Gal chl
Garrulus glandarius	Gar gla
Hippolais polyglotta	Hip pol
Cetropis daurica	Cet dau
Hirundo rustica	Hir rus
Lanius meridionalis	Lan mer
Lanius senator	Lan sen
Lullula arborea	Lul arb
Luscinia megarhynchos	Lus meg
Merops apiaster	Mer api
Miliaria calandra	Mil cal
Motacilla alba	Mot alb
Motacilla cinerea	Mot cin
Nycticorax nycticorax	Nyc nyc
Muscicapa striata	Mus str
Oenanthe hispanica	Oen his
Oriolus oriolus	Ori ori
Cyanites caeruleus	Cya cae
Lophophane cristatus	Lop cri
Parus major	Par maj
Passer hispanolensis	Pas his

Passer domesticus	Pas dom
Phoenicurus ochrus	Pho och
Phylloscopus ibericus	Phy ibe
Picus viridis	Pic vir
Ptyonoprogne rupestris	Pyt rup
Regulus ignicapillus	Reg ign
Saxicola rubicola	Sax rub
Serinus serinus	Ser ser
Sitta europea	Sit eur
Streptopelia turtur	Str tur
Streptopelia decaocto	Str dec
Sturnus unicolor	Stu uni
Sylvia atricapilla	Syl atr
Sylvia cantillans	Syl can
Sylvia conspicillata	Syl con
Sylvia melanocephala	Syl mel
Sylvia undata	Syl und
Sylvia spp.	Sylvia sp
Tringa ochropus	Tri och
Troglodytes troglodytes	Tro tro
Turdus merula	Tur mer
Turdus viscivorus	Tur vis
Upupa epops	Upu epo

u évora
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