MASTER 2 BIODIVERSITY, ECOLOGY AND EVOLUTION
ECOLOGY AND ECO-ENGINEERING OF WETLANDS



Utagawa Hiroshige (Ando) (Japanese, 1797-1858). Dam on the Otonashi River at Oji, No. 19 in One Hundred Famous Views of Edo, 2nd month of 1857.

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MASTER 2 INTERNSHIP REPORT

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ARTIFICIAL WETLANDS AND WATERBIRDS

A LITERATURE REVIEW

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1. Introduction

Wetland ecosystems are not only crucial for the survival of their component species, but also for the storage and provision of clean water for human use (Millennium Ecosystem Assessment 2005; Walsh et al. 2014). Indeed, wetlands are a key factor in the global water cycle and in regulating local water availability and quality through denitrification and detoxification. Wetlands also contribute to nutrient cycling, sediment transfer and flood protection (Zedler et Kercher 2005; Russi et al. 2013).

The importance of wetland ecosystem functions is often overlooked to such an extent that they were historically considered as 'wastelands' suitable for conversion to alternative uses (Vié et al. 2009). Consequently, many wetlands have been drained and converted for more 'profitable' uses through land use changes such as increased urbanization and conversion into agricultural land (Fraser et Keddy 2005). It is estimated that over 50% of the world's natural wetlands have disappeared (Russi et al. 2013), currently covering only 0.9 % of the Earth's surface (Costanza et al. 2014).

Globally, rapidly growing human populations are putting ever-greater pressure on the services performed by wetlands. Due to these past and ongoing demands on land and water, they are considered as one of the most threatened habitats worldwide (Szabo et Mundkur 2017; Mitsch et Gosselink 2015). Given this, the survival of many wetland-dependent species is becoming more precarious. Indeed, natural wetlands provide a wide variety of habitats, favoring a high level of biodiversity (Sarra et al. 2020). They also provides abundance of food sources to various organisms including waterbirds (Rahman et Ismail 2018). In general, waterbirds have become an important component of wetland environments (Canepuccia et al. 2007). They use them for feeding, roosting and if favorable would establish and breed in the area (Wetlands International 2012; Ramsar 2016). Hence, waterbirds are widely used as bio-indicators to evaluate the ecological status of the wetland habitats in which they are found (Burger 2006; Newman et al. 2007; David Li et al. 2009). It is widely acknowledged that the number of waterbirds using a wetland site is a good indicator of its biological importance (Scott 1980).Furthermore, migratory waterbirds highly depend on wetlands, notably as stopover sites. For instance, at least 42% of migratory waterbird populations are declining due to the loss of wetlands along major flyways area (BirdLife International 2012). In being ecologically dependent on wetlands for survival, reproduction, and recruitment, waterbird populations are declining worldwide (Nebel et al. 2008; Murray et al. 2014).

While natural wetlands continue to decline rapidly in quantity and quality (Davidson 2014), artificial wetlands have expanded in area, often at the cost of the conversion of natural wetlands (Bai et al. 2018). Natural wetlands converted between 1975 and 2005 in the Mediterranean Basin (for the 302 studied wetland sites) were transformed into equal parts of croplands and artificial wetlands (MWO2, 2018). It is worth noting that these latter increased by over one-quarter (26%) since 1970, with nearly two-thirds of them developed in natural wetlands such as wet meadows and riverine woodlands. Following the Ramsar

Convention, we considered as man-made wetlands aquaculture, farm ponds, and permanently or temporarily inundated agricultural land - such as rice paddies, salt pans, reservoirs, gravel pits, sewage farms and canals (Ramsar Convention Secretariat, 2011).

Due to habitat loss, waterbird species are obligated to use artificial wetlands as alternative habitats (Rajpar 2013), which are becoming increasingly important for their conservation (Toral et Figuerola 2010; Ramírez et al. 2012; Afdhal et al. 2013; Navedo et al. 2013; Yu et al. 2017). But not all species are able to use artificial wetlands (Toral et Figuerola 2010). Indeed, when natural habitat loss occurs, species differ enormously in their responses and the extent to which populations show negative trends following habitat loss strongly depends on their efficiency to alter their foraging behavior in order to exploit alternative food resources (Masero 2003).

Although it is logical that priority must be given to the conservation of natural wetlands (Donglai Li et al. 2013; Sebastián-González et Green 2016), understanding how waterbird communities are responding to changes in the wetland landscape, and to which extent artificial wetlands may compensate for the degradation of natural ones as waterbird habitats have become questions of great concern.

My objective is to synthesize existing information on waterbirds utilizing artificial wetlands to understand the importance of these ecosystems to waterbirds conservation and to identify areas in which additional research is needed. I expect that artificial wetlands provide less suitable habitat for waterbird communities. The latter are generally less abundant and composed of fewer species than those using natural wetlands (Tourenq et al. 2001; Donglai Li et al. 2013). Man-made wetlands are therefore less likely to support high levels of waterbird diversity than natural wetlands. However, they can in some cases play a crucial complementary role in conserving biodiversity. Additionally, I hypothesize that depth, size, connectivity, susceptibility to disturbance, and the presence of both emergent and adjacent vegetation are all factors that will probably affect avian diversity in artificial wetlands (C. G. Murray et al. 2013; Bino et al. 2015; Giosa et al. 2018).

2. Materials and methods

2.1. Search strategy

In this paper, I chose the Web of Science database (https://www.webofknowledge.com/) to explore the literature associated with the use of artificial wetlands by waterbirds. I used the following combination of keywords and operators: artificial* AND (waterbird$ OR shorebird$ OR seabird$ OR duck$ OR waterfowl), with the "*" allowing for additional letters, and "$" allowing for a single additional letter. This expression was only searched in publication titles to avoid impertinent results. These requirements were matched by 78 articles. I did not set any timespan to limit my search.

Among the 78 articles, I rejected studies outside my scientific field, such as behavioral ecology. I also completed my search with other relevant papers cited in the sources that were selected with the systematic search.

3. Results

3.1. Reduction of avian diversity in artificial versus natural wetlands

According to several studies, the abundance and species richness of waterbirds are lower in artificial wetlands in comparison with natural ones (Ma et al. 2004; Donglai Li et al. 2013; Sebastián-González et Green 2016; Giosa et al. 2018). Moreover, findings of various research suggest that artificial wetlands tend to favour some waterbird species over others (Bellio et al. 2009; Donglai Li et al. 2013), and may therefore not perform all the functions of natural wetlands.

Donglai Li et al. (2013) showed that many avian community measures (e.g., species richness, relative abundance and mean diversity) were higher in natural wetlands of the Yellow River Delta (Eastern China) than in the neighboring artificial wetlands. Furthermore, natural wetlands provided suitable habitat for almost all waterbird guilds, i.e., pelicans, cormorants and grebes, cranes, coots, large-bodied waterbirds, terns, gulls, waterfowl and shorebirds. However, there were exceptions to this trend, as aquaculture ponds hosted the most species of storks, egrets, herons, cranes and shorebird guilds. This suggests that although some waterbird guilds may benefit from a uniform artificial wetland, the latter will greatly erode the avian diversity (Navedo et al. 2012).

This mirrors the findings of other waterbird research that documented specific waterbird assemblages depending on wetland types (e.g. Bellio et al. 2009; Almeida et al. 2020), and possibly reflecting ecological resource variations that each habitat provided. For instance, in south-east Sri Lanka, waterfowl species preferred irrigation tanks over other neighboring types of wetlands (i.e. salt ponds, rice paddies, coastal lagoons), whereas shorebird species were more abundant on coastal lagoons and salt evaporation ponds (Bellio et al. 2009). However, it should be noted that artificial wetlands could not replace the conservation value of the natural coastal lagoons in supporting a rich waterbird community, particularly migratory shorebirds. Thus, compensating the loss of natural wetlands with artificial wetlands leads to a lower avian functional diversity (Ma et al. 2004; Sebastián-González et Green 2016; Almeida et al. 2020).

Furthermore, as shown in previous studies, artificial wetlands are less likely to support threatened species (Donglai Li et al. 2013; Sebastián-González et Green 2016; Jackson et al. 2020). Having said that, any assessment of the conservation value of an artificial wetland has to be context dependent. Human-made wetlands can sometimes be of considerable value for some species of cultural or conservation interest (e.g. the endangered White-headed Duck (*Oxyura leucocephala*) and the vulnerable Marbled duck (*Marmaronetta angustirostris*) found in artificial wetlands in Cyprus (Giosa et al. 2018).

As artificial wetlands seem not adequate to replace natural wetlands, it would be more reasonable to prioritize the conservation of natural wetlands as they tend to have higher habitat heterogeneity and can therefore accommodate more waterbird species (Wang et al. 2020) and more individuals (Giosa et al. 2018). However, this does not negate the fact that under the appropriate conditions, certain artificial wetlands seem capable of providing habitat to diverse communities (Cardoso et Loures-Ribeiro 2017; Rajpar 2013). The factors making an artificial wetland more likely to support diverse waterbird communities are detailed in the next section.

3.2. Artificial wetland attributes influencing community structure of waterbirds

The factors making an artificial wetland suitable for waterbirds are multiple (C. G. Murray et al. 2013). Several studies have documented that the size of artificial wetlands influences abundance and richness of waterbirds (Froneman et al. 2001; Ma et al. 2004; Sánchez-Zapata et al. 2005; Afdhal et al. 2013; Giosa, et al. 2018). Indeed, larger wetlands tend to support more species as they usually offer higher habitat heterogeneity and hence more foraging and breeding habitats than smaller wetlands (Paracuellos 2006). Furthermore, artificial wetlands of small size are susceptible to disturbance from the nearby area (Ma et al. 2000). This implies that more waterbirds would occur in large wetlands [e.g., farm ponds in Froneman et al. (2001); artificial ponds in Sánchez-Zapata et al. (2005) and Sebastián-González et Green (2013); dams in Afdhal et al. (2013)].

Among the hydrologic variables within wetland ecosystems, water depth has been demonstrated to affect waterbird diversity (Bancroft et al. 2002; Webb et al. 2010; Faragó et Hangya 2012; Giosa et al. 2018; Ionescu et al. 2020). Indeed, different species prefer different depths even within foraging guilds like ducks (Murakani et al. 2015). Moreover, it is well documented that shallow wetlands, offer different depth needs for a large spectrum of waterbird groups (e.g., 10-20 cm, Isola et al. 2000; Colwell et Taft 2000), and thus accommodate more species and greater numbers of waterbirds than deep wetlands. Water depth is known to affect the access to foraging habitats (Murakani et al. 2015) but also the net energy intake of waterbirds (Ma et al. 2010). For instance, Gawlik (2002) showed that for wading birds that forage in the water column, deeper water can reduce their foraging efficiency since prey can escape not only horizontally, as is the case in shallow areas, but also vertically. The foraging efficiency reduction is also valid for the non-diving herbivores. Murakani et al. (2015) demonstrated that herbivorous waterfowl species preferred shallow areas less than 5m in depth. This was explained by the light limitation that restricted the distribution of underwater vegetation to shallow areas. This suggests that a mosaic of small and large ponds of variable depths could benefit different waterbird groups (Sebastián-González et Green 2013).

Additionally, water salinity has been identified as an important hydrologic variable explaining waterbird diversity within a wetland (Brochet et al. 2009). Indeed, salinity affects the community composition of aquatic plants and consequently, affects the distribution of aquatic animals and zoobenthos (Ma et al.

2010). This indirectly influences the waterbird use of wetlands as foraging sites. The effects of water salinity on different invertebrates are taxa dependent. Therefore, the optimal water salinity at foraging sites varies with waterbird diet composition. For instance, Almeida et al. (2020) showed that large wading birds, such as purple heron (*Ardea purpurea*), associated with low saline wetlands, are absent in artificial fish ponds. This group prey on large insects, amphibians and other species absent in the high saline habitats such as fish ponds. On the other hand, the greater flamingo (*Phoenicopterus roseus*) occurred in high abundances in saline habitats. All these results suggest that, when managing saline environments for waterbirds, salinity must be regulated according to the prey composition of waterbird diets in order to support foraging habitats for diverse species.

Another crucial parameter that greatly influences the habitat use of waterbirds is vegetation (Ma et al. 2010). Apart from being an important food source for herbivorous species, it can provide shelter and reduce human disturbance especially in artificial wetlands, at both nesting and roosting sites (Hattori et Mae 2001; Sánchez-Zapata et al. 2005). Many studies have documented that species abundance and richness increase with increasing emergent vegetation cover in wetlands, particularly during the breeding season when waterbirds are more sensitive to disturbance and less mobile (Froneman et al. 2001). However, most waterbirds cannot stand on extremely high or low vegetation for foraging. Moderate vegetation heights, however, offer both perching and shelter (Bancroft et al. 2002). Yang et al. (2016) showed that the highest waterbird species density and richness were recorded in mix wetlands, where high heterogeneity of vegetation heights occurred.

Other variables related to water quality (e.g., dissolved oxygen, pH, temperature) (Barragán Severo et al. 2002), the characteristics of sediments (Mwaura et al. 2002) and the land use around artificial wetlands (Sebastián-González et Green 2013) also affect the waterbird habitat use. For instance, the amount of organic matter in water and sediments affects the development of aquatic vegetation and thus determines invertebrate abundance (Rehfisch 1994). Bellio et al. (2009) have suggested that increased sediment loads from agricultural runoff contribute to higher turbidity on the Embilikala coastal lagoon (SE Sri Lanka), and negatively affect aquatic benthic invertebrates. This led to lower densities of visual foraging shorebirds, that feed almost entirely on benthic invertebrates. The same was found in the Santana Reservoir, located in Rio de Janeiro, Brazil (Silva et al. 2020). Due to its eutrophic status, it provided large food reserves (i.e., macrophytes) for the gleaning bird species such as *Gallinula galeata* and *Jacana jacana* but disadvantaged the visual foraging fishing birds, which displayed low local abundance. Since they are visual foragers, they highly depend on the macrophyte-free water and low turbidity levels to capture prey.

Few papers mentioned the relationship between the effectiveness of protected areas and waterbird species diversity within artificial wetlands. Indeed, waterbird hunting is known to cause disturbance and displacement of waterbirds in many wetlands (Brochet et al. 2009). The creation of nature reserves is a crucial and widely used measure to reduce disturbance effects. In Cyprus, where hunting is a popular

activity, Giosa et al. (2018) showed that the percentage of the area covered by reserves where hunting is prohibited had a positive effect on waterbird species richness in the nineteen studied wetlands (both natural and artificial). Similarly, Ionescu et al. (2020) have demonstrated that waterbird colonies (herons and egrets) prefer the artificial ponds within protected areas, where integrated water management is applied, during the breeding season.

All these suggest that wetland characteristics may be more crucial in determining species diversity within sites than the wetland type itself. Because of their higher resource predictability and seasonal stability when compared to natural wetlands (Márquez-Ferrando et al. 2014), man-made wetlands can play a complementary role in conserving waterbirds species. This aspect is discussed in more depth in the following section.

3.3. Complementary use of natural and artificial wetlands

In natural wetlands, resource availability is strongly variable and highly depends on rainfall and climate regime which changes both seasonally and annually (Serrano et al. 2006). In contrast, artificial wetlands such as fishponds, salt ponds or rice fields can provide waterbirds with a stable and relatively safe environments (Márquez-Ferrando et al. 2014) and may even be preferred as feeding habitats at specific times of the year (Tourenq et al. 2001).

A good example to illustrate this is the Doñana wetlands in the estuary of the Guadalquivir River in south-western Spain. According to Rendón et al. (2008), Doñana is one of the most important wetlands in Europe for wintering and migrating waterbirds along the East Atlantic flyway. Actually, at the start of the 20th century, both sides of the river were covered by vast natural marshes (more than 100 000 ha), which were transformed into rice-fields, saltpans and extensive aquaculture in the 1990s. As a result, a complex mosaic of diverse aquatic habitats from natural to artificial wetlands is found (Ramírez et al. 2012). Previous research showed that many waterbird species use artificial and natural wetlands in Doñana in a complementary way during winter (Kloskowski et al. 2009; Márquez-Ferrando et al. 2014). For instance, Kloskowski et al. (2009) suggested that the proportion of wintering Anatidae species present in the nearby fish farm of Veta la Palma, a permanently flooded man-made system, was much higher during dry winters when the natural marshes are not flooded than in wet winters. Also, ducks redistributed to the natural marshes as more of them were flooded. This system (i.e. the combination of natural marshes and artificial fishponds) allows the complementation of resources which enables the Doñana wetland complex to support a larger and more diverse community of wintering waterbirds than if the entire site had only a natural marshland. Here, the artificial fishponds function as buffer areas during drought periods, reducing population declines.

The same result was found for some waterbirds species using fishponds during the breeding season, such as greater flamingos (*Phoenicopterus roseus*) in the Fuente de Piedra lake, Malaga (Amat et al. 2005) and slender-billed gulls (*Chroicocephalus genei*) in the Natural Area of Doñana (Ramírez et al. 2012).

Another illustration of the complementary use of natural and artificial wetlands is the utilization of rice fields as foraging sites during the breeding season. Indeed, in the Northern Hemisphere, fields are flooded from April to September, which coincides with the breeding season of many waterbirds (Thibault et al. 1997; Tourenq et al. 2001; Czech et Parsons 2002). The Audouin's Gull (Larus audouinii) for instance, a marine predator that benefits from scavenging fish discarded from trawlers, has become a species that exploit rice fields due to changes in fisheries management policies (Ruiz et al. 1996). In the provinces of Tarragona and Castello in Spain, trawl fishing moratoriums were instated during the gull breeding season in order to counteract the effect of overfishing on stocks. Consequently, gulls increased foraging in rice fields to complement prey caught in pelagic habitats although it provided less biomass than could be gained from scavenging (Oro et al. 1996). Similarly, in the Camargue, rice fields are selected by purple herons (*Ardea purpurea*) in autumn. Since temporary natural marshes are often dry at the end of the summer (Tourenq et al. 2001), rice fields provide alternative foraging habitats for this migrant species, necessary to fuel their migration.

Furthermore, in urban areas, reservoirs can provide valuable habitat for waterbirds by providing roosting, feeding and breeding habitat for resident waterbirds (Silva et al. 2020). Indeed, during the dry season, when surrounding natural wetlands could fragment and dry, higher abundances were recorded on available artificial wetlands such as water-level controlled reservoirs in Rio de Janeiro, Brazil (Silva et al. 2020) and large dams in the Murray-Darling Basin, eastern Australia (Bino et al. 2015), as they can serve as a drought refuge and provide resources for waterbirds to survive.

The complementary use of natural and artificial wetlands by waterbirds suggests that an integrated landscape with multiple wetland types can meet the habitat requirements of various waterbird species and thus support high diversity. This is consistent with previous research that has shown that artificial wetlands mostly built for economic purposes can also be used as supplementary or alternative habitats for waterbirds (Green et al. 2015; Jackson et al. 2019).

4. Discussion

As highlighted before, natural wetlands are among the most threatened habitats worldwide (Fraser et Keddy 2005; Russi et al. 2013; Costanza et al. 2014). The threats include urbanization, conversion to agricultural uses and to artificial wetlands (Ramsar, 2018). As a result, waterbirds are suffering population declines globally (Kirby et al. 2008). Moreover, they face serious threats related to climate change, through

loss of habitat, driven by changed conditions on the breeding grounds and sea level rise. In this context, artificial wetlands are playing an important role in maintaining avian diversity by providing favorable foraging, roosting and breeding sites for some species (Silva et al. 2020). Although these human-made habitats may only act as refugia habitats to waterbirds in time of food scarcity (Bino et al. 2015), their conservation value become increasingly apparent in order to mitigate the influences of degradation of natural wetlands (Zedler et al. 2000; Czech et Parsons 2002).

In this scenario, a detailed understanding of the full extent of natural and artificial waterbird habitat use is highly required. In fact, the scientific literature reviewed in the present paper is lacking the following research questions: Firstly, investigation to what extent waterbird diversity is supported by artificial wetlands at a larger spatial scale. This would allow us to cover a larger spectrum of artificial wetland types (reservoirs, salt ponds, rice fields, fishponds, etc.) compared to the numerous studies that generally focus on a single artificial wetland type (e.g., dams in Northern Tunisia (Hamdi 2014); coastal artificial wetlands in the Asia-Pacific (Jackson et al. 2020); rice fields in the Camargue (Tourenq et al. 2001); urban water reservoirs in Rio de Janeiro (Silva et al. 2020)). Additionnaly, surveys taking into account both wintering and breeding ranges of waterbird species by exploiting international censuses is also needeed.

Secondly, surveys comparing waterbird communities before and after the anthropization of wetlands are very much needed to understand how it affects community composition. For instance, the shift from lotic to lentic environments after dam construction have significant effects on waterbird community composition (Qing et al. 2017). Currently, there is a major resurgence in dam construction, particularly in countries with emerging economies such as Anatolia and the Balkans (MWO2, 2018). Hence the urgent need to understand to what extent waterbird functional diversity is eroded by hydropower dam construction. This may allow us to ensure that dams are designed and managed in a way that limits their impact on the biodiversity associated to these habitats.

Finally, additional studies should be programmed taking into account the biophysical features of wetlands and the nature of the surrounding landscape that need to be incorporated into development plans and artificial wetland construction processes.

However, it is important to remind that over-emphasizing the role of artificial wetlands in conserving waterbird diversity may inappropriately encourage landowners to convert natural wetlands to other uses, thus accelerating the loss of natural habitats. It is here suggested that the conservation value of artificial wetlands should not be discounted in a context of increasing loss of natural habitats due to land conversion and climate change. Although it is judicious that conservation actions should prioritize natural wetlands, it can be complemented by the protection of biodiverse artificial wetlands. Regardless of their original construction aim, which is principally human activities, managing artificial wetlands as a complement to

natural habitat preservation, is crucial for waterbird conservation, especially when no natural wetlands are available.

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CHARACTERIZING THE COMPOSITION OF WINTERING WATERBIRD COMMUNITIES OF DAM RESERVOIRS IN THE MEDITERRANEAN BASIN

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I- Introduction

Worldwide, rapidly growing human populations are putting ever-greater pressure on wetlands and the services they provide. Indeed, past and ongoing demands on land and water resulted in wetlands being one of the most threatened habitats globally (Mitsch et Gosselink 2015; Szabo et Mundkur 2017). In this context, Mediterranean wetlands are of major concern, facing a 45% loss since 1970, with noteworthy multifunctional natural wetlands being irreversibly converted (MWO2 2018). Across the basin, wetland sites are directly impacted by the increase of land-use change, driven for the most part by agriculture, urbanization, but also tourism (CEPF 2017; Leberger et al. 2020). Wetland loss and degradation in the region might by exacerbated by other global change components such as climate change (MedECC 2020). Indeed, the Mediterranean region is one of the most prominent climate change hotspots across the globe which makes it highly vulnerable to global change (Giorgi 2006). Drought episodes are expected to become more frequent and tenacious (Mariotti et al. 2015; Piero Lionello et Scarascia 2018; Raymond et al. 2019). Also, an increase in the number of heat waves in the upcoming decades is very likely (IPCC 2013).). These aspects of climate change are likely to raise serious impacts on wetlands and will probably lead to important changes in the region's water cycle (Lionello 2012). On the other hand, the Mediterranean basin is recognized as a biodiversity hotspot (Mittermeier et al. 2005; CEPF 2017) with very high rates of endemism (MWO2 2018), and where biodiversity is among the most sensitive to land-use modifications and climate change on the planet (Newbold et al. 2020).

Given this, the survival of many wetland-dependent species is becoming more precarious. Indeed, wetlands provide a wide variety of habitats, favoring a high level of biodiversity (Sarra et al. 2020). They also provide food sources to various organisms including waterbirds (Rahman et Ismail 2018) that have become an important component of wetland environments (Canepuccia et al. 2007). They use them for feeding, roosting and if favorable would establish and breed in the area (Wetlands International 2012; Ramsar 2016). Hence, waterbirds are widely used as bio-indicators to evaluate the ecological status of the wetland habitats in which they occur (Burger 2006; Newman et al. 2007; David Li et al. 2009). Partly because they are ecologically dependent on wetlands, waterbird populations are declining worldwide due to habitat loss (Nebel et al. 2008; Murray et al. 2014). Although the overall population trend has been estimated as positive within the Mediterranean basin, this trend is mainly driven by western populations in contrast with the decreasing eastern populations (Galewski et al. 2011).

While natural wetlands continue to decline rapidly in quantity and quality (N. Davidson 2014), artificial wetlands have expanded in area, often as a result of the conversion of natural wetlands (Bai et al. 2018). Indeed, the global surface area of human-made wetlands has increased by over one-quarter (26%) since 1970, with the largest increase documented in the southeastern Mediterranean basin (MWO2 2018). This was coupled with an increase of available surface water due to the construction of dams and their associated water impoundments (Pekel et al. 2016), most of which are inland rather than coastal (Davidson

et al. 2018). Although artificial water bodies such as reservoirs can partly substitute natural water bodies in the total available habitat for waterbirds (Leberger et al. 2020), most species do not benefit from artificial wetlands and are decreasing, such as migratory fish (Lucas et al. 2009), amphibian larvae (Dare et al. 2020) and other lotic organisms (Galewski et al. 2021). Yet, given the heavy loss of natural wetlands, waterbird species are forced to use artificial wetlands as alternative habitats (Rajpar 2013), making them increasingly important for avian conservation (Toral et Figuerola 2010; Ramírez et al. 2012; Afdhal et al. 2013; Navedo et al. 2013; Yu et al. 2017). Waterbird communities in man-made wetlands are generally less abundant and composed of fewer species than those in natural wetlands (Tourenq et al. 2001; Donglai Li et al. 2013). However, not all species are able to use artificial wetlands (Toral et Figuerola 2010). Indeed, when natural habitat loss occurs, responses differ greatly among species, and the extent to which populations show negative trends following habitat loss strongly depends on their efficiency to alter their foraging behavior in order to exploit alternative food resources (Masero 2003).

Given the current major resurgence in dam construction within the Mediterranean region, particularly in countries with emerging economies such as Anatolia and the Balkans (MWO2, 2018), there is an urgent need to understand to what extent waterbird species are supported by these man-made habitats. Hence, we evaluate here wintering waterbird community composition within dam reservoirs for 22 Mediterranean countries between 1993 and 2017. From the International Waterbird Census counts and the georeferenced Reservoir database, a dam data source established by the Tour du Valat, that repertories the largest number of dams at the scale of the Mediterranean Basin, we (1) characterized dam reservoirs that encompassed waterbird surveys using a set of environmental variables, (2) identified which environmental factors affected waterbird associations within wetland sites, (3) tested the dissimilarities in both species and functional group composition between dam sites and other wetland types, and (4) investigated if the waterbird composition dissimilarities were climate dependent.

II- Materials and methods

1. Study area

Our study area encompassed 22 Mediterranean countries included in the Mediterranean Wetlands Initiative (MedWet, https://medwet.org/fr/aboutwetlands/ramsarmedsites/) that were also contracting parties to the Ramsar Convention and regularly contributed to the International Waterbird Census (see below): Albania, Bosnia and Herzegovina, Bulgaria, Croatia, Cyprus, Egypt, France, Greece, Israel, Italy, Jordan, Lebanon, Macedonia, Malta, Morocco, Montenegro, Portugal, Serbia, Slovenia, Syrian Arab Republic, Tunisia, and Turkey. Spain, Algeria, and Libya were not included in our study due to a lack of available data on georeferenced dams.

We grouped the study countries into four sub-regions according to the geo-political context (MWO2, 2018): Maghreb (Morocco and Tunisia), Middle East (Cyprus, Egypt, Israel, Jordan, Lebanon, Turkey, and Syria),

Balkans (Albania, Bosnia & Herzegovina, Bulgaria, Croatia, Macedonia, Montenegro, Serbia and Slovenia) and Southern Europe (France, Greece, Italy, Malta and Portugal).

2. Waterbird surveys

International Waterbird Census (IWC) counts were completed in mid-January in each wetland site by professional ornithologists or volunteers and coordinated globally by Wetlands International (see Delany (2005) for detailed count protocol). The IWC aims to monitor wetland sites and the waterbird populations they support in order to assess their size and trends during the wintering period, which encompasses January for most species. It should be noted that the count data from North-African countries (Morocco, Tunisia, and Egypt) were directly transferred by national coordinators.

IWC count data were filtered before proceeding to data analysis (F. Verniest, comm.pers). First, species that didn't match the definition of a waterbird as mentioned in the IWC protocol (Delany 2005), as well as domestic and hybrid waterbird species were excluded. Next, we aggregated subspecies observations to their parent species and deleted counts that fall outside a 15-day window around January 15. Also, we excluded observations for which more than 10 percent of the individuals were unidentified at the species level. To avoid vagrant species, we removed data for the 60 waterbird species outside the African-Eurasian migratory flyway based on waterbird population estimates from Wetlands International (2012). Following Gaget et al. (2020), we filtered out waterbird species with a wintering distribution range outside our study area based on BirdLife maps (BirdLife International, 2017). As in Popoff et al. (2021), *Larus michahellis*, *Larus cachinnans*, *Larus armenicus* and *Larus argentatus* were lumped together as Larus '*argentatus*' due to a taxonomy progress that occurred late in the survey period. Finally, IWC sites without coordinates, with inaccurate coordinates, or that may correspond to a complex of sub-sites were deleted.

We thus analyzed a database of 44 041 wetland surveys between 1993-2017 of 151 waterbird species monitored in 3813 sites (Fig. 1). The species belonged to 18 families and 9 orders.

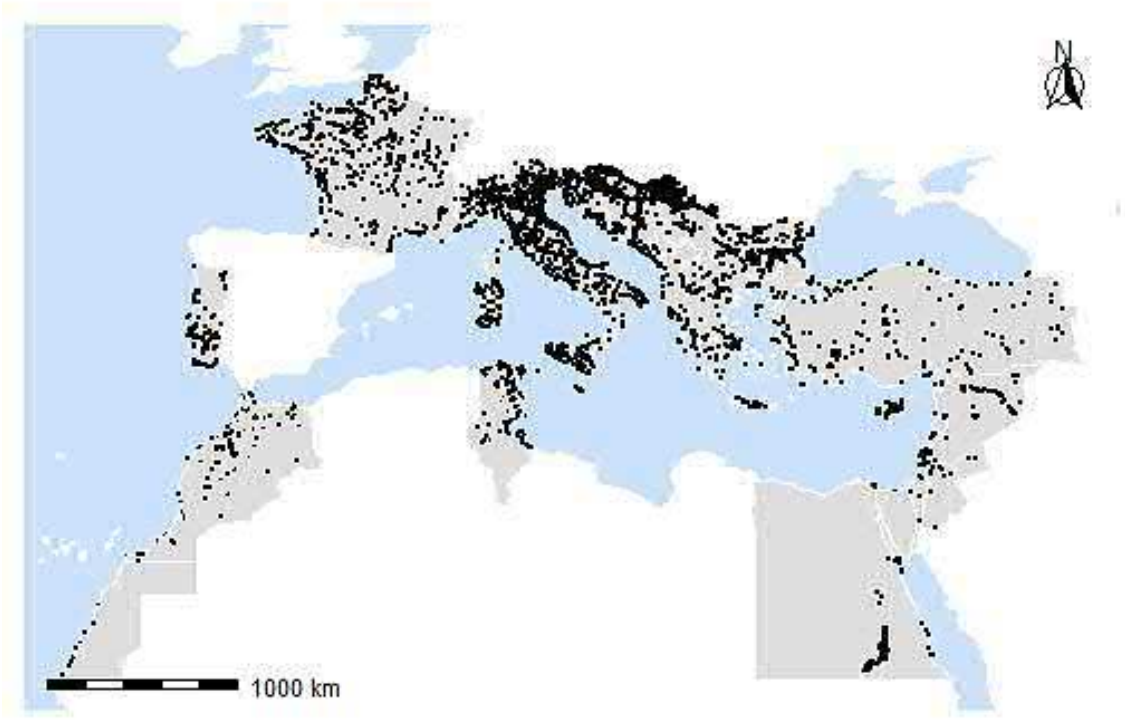


FIGURE 1 Distribution of the 3813 IWC sites (black dots) located in the 22 MedWet countries.

3. Waterbird conservation status

The conservation statuses of waterbirds were evaluated according to BirdLife International European categorization and qualified as Species of European Conservation Concern when their status was unfavourable (see below): SPEC 1 (European species of global conservation concern), SPEC 2 (species with global population concentrated in Europe and with an unfavourable conservation status in Europe) and SPEC 3 (species not concentrated in Europe, but with an unfavourable conservation status in Europe) (BirdLife International 2017). Species with a favourable conservation status were mentioned as 'Secure'. This classification was preferred over the IUCN Red List because it was updated more recently and is defined at a finer spatial scale when considering the study area (see Appendix 1 for details).

4. Functional groups

Waterbird species were grouped into functional groups according to their feeding habits. We considered 10 groups - of which 8 were similar to those defined by Almeida et al. (2020) for a more restrained geographic area : small wading birds (SWB), large wading birds (LWB), vegetation gleaners (VGI), dabbling ducks (DbD), diving ducks (DvD), opportunistic gulls (OpG), surface fishers (SfF), and diving fishers (DvF). We also added 2 groups of species that were not considered in Almeida et al. (2020): dabbling geese (DbG) and dabbling swans (DbS) (see Appendix 1).

5. Dam data

We used the most comprehensive and up to date source of georeferenced dam reservoirs established by the Tour du Valat at the scale of the Mediterranean Basin (OZMH 2020). This database was constructed using international dam databases, notably the Global Reservoir and Dam database (GranD) and FAO AQUASTAT data. It was complemented by means of satellite image interpretation and grey literature mining. 34 516 dam reservoirs were identified in the study area (see Appendix 2 for the area covered by dams per country). The database provides information on dam name, country, and reservoir area. Note that the dam database repertoried the largest number of dams, with sizes ranging from less than 1 km² to over 5000 km², compared to other international georeferenced dam databases such as the GranD that include only a subset of existing dams, mostly those with a reservoir storage superior to 0.1 km3 (Lehner et al. 2011). For instance, the GRanD database recorded only 5 dams in Albania, against 545 for our dam database.

To determine the number of dam reservoirs that enclosed waterbird surveys, we overlaid dam locations with points and polygons from the International Waterbird Census (IWC) counts.

6. Mediterranean sites protection

To assess whether the IWC sites fell within protected areas, we used the most up-to-date database of protected areas in the Mediterranean region made in collaboration between the Tour du Valat and Wetlands International (OZMH 2020). This database uses several data sources, including World Database on Protected Areas (www.protectedplanet.net), Natura 2000 (www.eea.europa.eu) and other national inventories. A site was considered as covered by a protected area when its geographical coordinates overlapped with a protected area polygon. The database provides information on site name, country, area, designation year and the IUCN category when available.

7. Environmental variables

We distinguished coastal from inland IWC sites to avoid misinterpretations of potential differences in waterbird community composition between dams and other wetland types. Following Popoff et al. (2021), an IWC site was considered as coastal when its centroid was ≤ 30 km from the coastline. This definition is exclusively geographic and thus does not necessarily imply that sites fall within coastal ecosystems. We defined climate classes of IWC sites and dams using the Köppen–Geiger climate classification provided by Beck et al. (2018) at a 5 arc-minutes spatial resolution (approx. 86 km2 at the equator). Each climate class is derived from monthly mean temperature and precipitations and described by a three-letter code.

The degree of human pressure on study sites was assessed using the human footprint (HFI) index as a proxy (Venter et al. 2016). HFI is a spatial index of cumulative human pressure ranging from 0 to 50, where 0 corresponds to areas free from human influence called ‘wilderness areas’, and values above 20 indicate

very high-pressure levels (see Venter et al. (2016) for detailed methodology). HFI was mean-aggregated to a 5 arc-minutes spatial resolution in order to match climate data resolution.

Altitude data was obtained from the Shuttle Radar Topography Mission (SRTM, Farr et al. 2007).

8. Data analysis

8.1. Characterization of the sampling dams and IWC sites

To ensure representativeness and improve comparability of waterbirds communities between IWC sites within dams and other wetland sites, we characterized both dam and IWC site within dams' samples. To this end, we first compared the dams that encompassed waterbird surveys with those that didn't, based on area, climate category, human footprint index (HFI) category, altitude, and protected area IUCN category when available. The same parameters were then investigated to compare IWC sites that fell within dams with those that didn't (other wetlands) except for the area.

An independent sample Wilcoxon rank-sum test for continuous variables (Wilcoxon 1992) was used to detect differences in area, altitude, and Human Footprint (HFI) between counted dams and not counted dams, and between IWC sites within dams and the rest of IWC wetland sites. For climate and protection (IUCN categories) differences, a Chi-Square Test of Independence was performed.

Note that we computed mean species abundances over the count period for each IWC site.

8.2. Differences in waterbird specific and functional composition

In this part, we first used a Canonical Correspondence Analysis (CCA, ter Braak 1986) to evaluate the influence of environmental variables (climate, Human Footprint, coast, site protection) on the association of waterbird species and on the association of functional groups. CCA is the canonical form of correspondence analysis (CA). It is an asymmetric constrained ordination analysis that identifies and measures associations among two sets of variables: species abundances and environmental gradients; it was often used in long-term ecological studies (Verniest et Greulich 2019). Through the CCA diagram, we also visualized potential dissimilarities in both species and functional group composition between dam sites and other wetland types.

Then, we tested these dissimilarities using a permutational multivariate analysis of variance (PERMANOVA, Anderson 2001). It is a non-parametric alternative to a multivariate ANOVA, allowing us to assess multiple dependent variables (i.e., species or functional groups) simultaneously. PERMANOVA was parametered to use the Bray-Curtis dissimilarity matrix and carried out using the adonis function from the "vegan" package (Oksanen et al. 2020). We ran 50000 permutations to ensure accurate P-value estimates.

The following variables were categorized before the permutational multivariate analysis (PERMANOVA): HFI was categorized using a quartile scale, the study countries were grouped into sub-regions (see Study area for details), protected sites were categorized into non-protected sites, sites within IUCN categories

ranging from I to IV and those within the remaining protection assignments. Environmental variables were then combined into a single variable called 'Envr': climate (14 groups, see Appendix 3 for details), HFI (4 groups), region (4 groups), IUCN I-IV (3 groups), coast (2 groups: yes/no). The dam effect was investigated with a binary variable ("yes" or "no"). Explanatory variables in the PERMANOVA included the 'Envr' and 'DAM' variables.

A second PERMANOVA was performed to evaluate to which extent waterbird composition differences between dams and the rest of IWC sites were climate dependent. To this end, we grouped the identified climates into three main types according to the Köppen–Geiger classification (Beck et al. 2018): arid, temperate, and cold. This updated climate variable was then used as an explanatory variable with the 'DAM' variable, their interaction, and the updated 'Envr' variable (without climate).

R software version 4.0.5 was used for all statistical analyses. Geoprocessing operations were performed on R and QGIS 3.4.15 (QGIS Development Team, 2020). Results were considered as statistically significant when p-value was below 0.05.

III- Results

1. *Characterization of the sample dams*

Only 465 out of 34 516 existing dams (1.35%) have been surveyed with an important regional heterogeneity of the IWC dam sampling effort within the study area (Fig. 2).

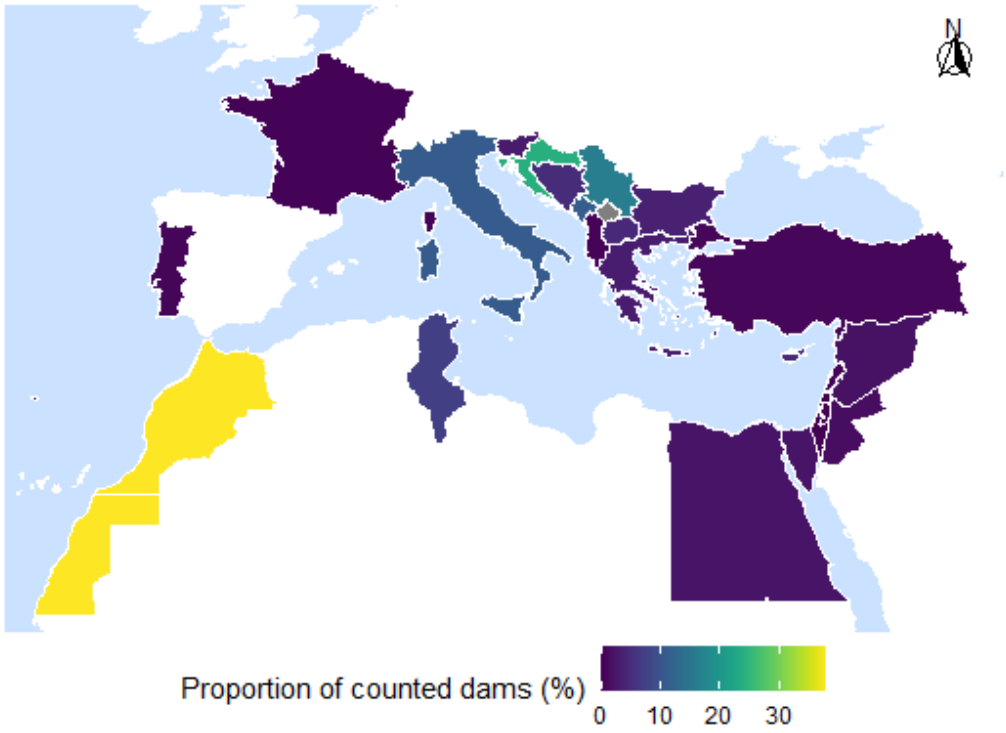


FIGURE 2 Counted dam concentration in the study area

1.1. Area

There were significant area differences between the dams that encompassed waterbird surveys and those that didn't (Wilcoxon rank-sum test, W= 14898472; $p < 0.001$). IWC counts were biased towards large dams (48.78% of dams with areas ranging from 100 to more than 1000 km² were surveyed), while smaller dams (with areas less than 1 km²) were the most common type in the study area but the least surveyed area category (see Fig. 3). Indeed, only 0.73% of small surface dams (241 out of a total of 33 128 dams) encompassed IWC counts.

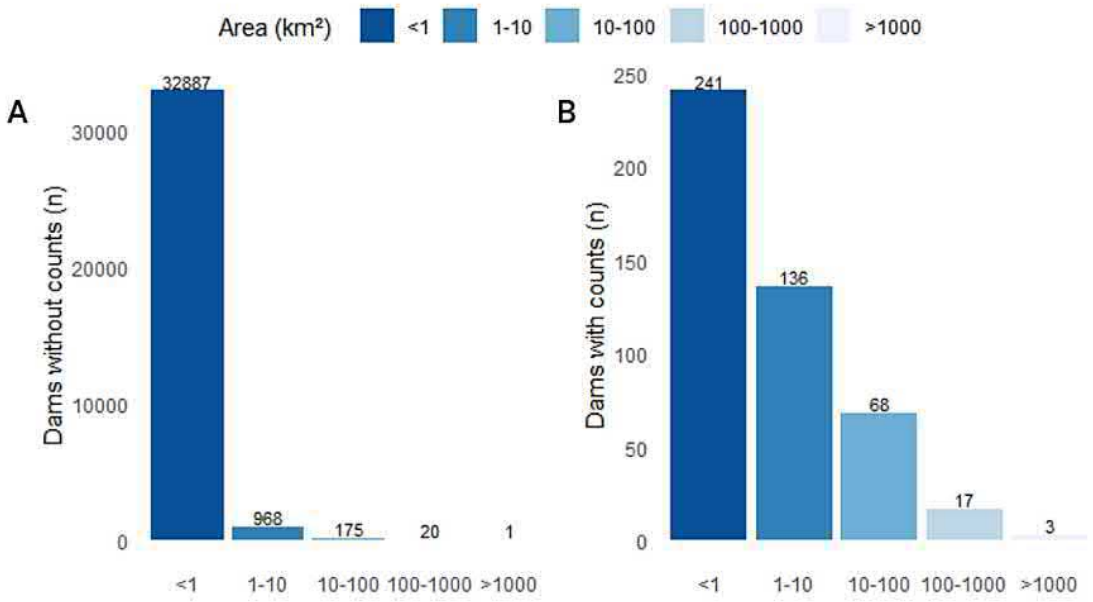


FIGURE 3 Dam numbers in each area category for both uncounted (A) and counted (B) dams.

1.2. Human Footprint

The Human Footprint Index (HFI) values were significantly different between the dams containing IWC counts and those that didn't (Wilcoxon rank-sum test, W= 9550975; $p < 0.001$), with higher values in the first category (mean=14.7; sd= 4.89 HFI) compared to those in the second category (mean= 13.3; sd= 5.32 HFI) (Fig. 4).

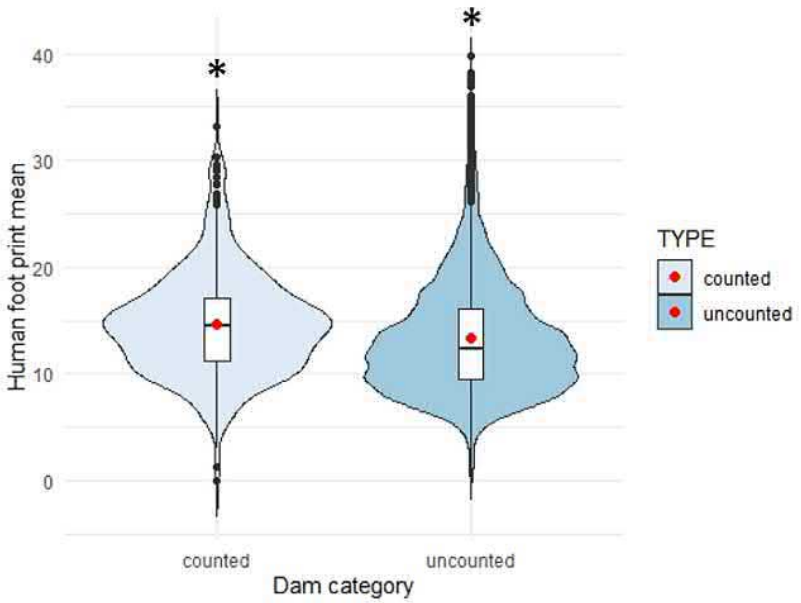


FIGURE 4 Distribution of the Human Footprint Index (HFI) across counted and uncounted dams.

1.3. Altitude

Counted dam altitude was higher (mean= 390m; sd= 375m) than that of the dams not containing waterbird counts (mean= 332m; sd= 366m, Fig. 5). These differences were statistically significant (Wilcoxon rank-sum test, W= 8841052; $p < 0.001$).

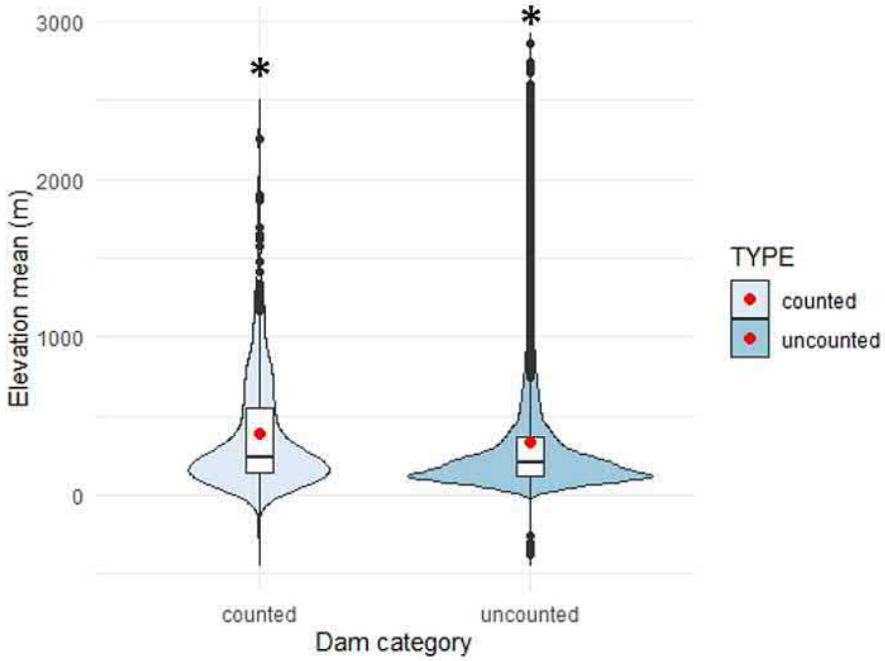


FIGURE 5 Distribution of mean altitude (m) across counted and uncounted dams.

1.4. Climate

The most common climate type within dams was temperate climate (84.23%), followed by cold climate (9.23%), arid climate (6.45%) and finally polar climate (0.08%) for both surveyed and non-surveyed dams (Fig. 6). Although temperate dam sites were the most counted category in the study area, they represented the lowest sampling effort if we ignore the polar climate, with only 0.96% dams containing waterbird surveys. The difference in the climate type distributions between dams that contained IWC counts and those that didn't was significant ($X^2 = 225.17$, $p < 0.001$).

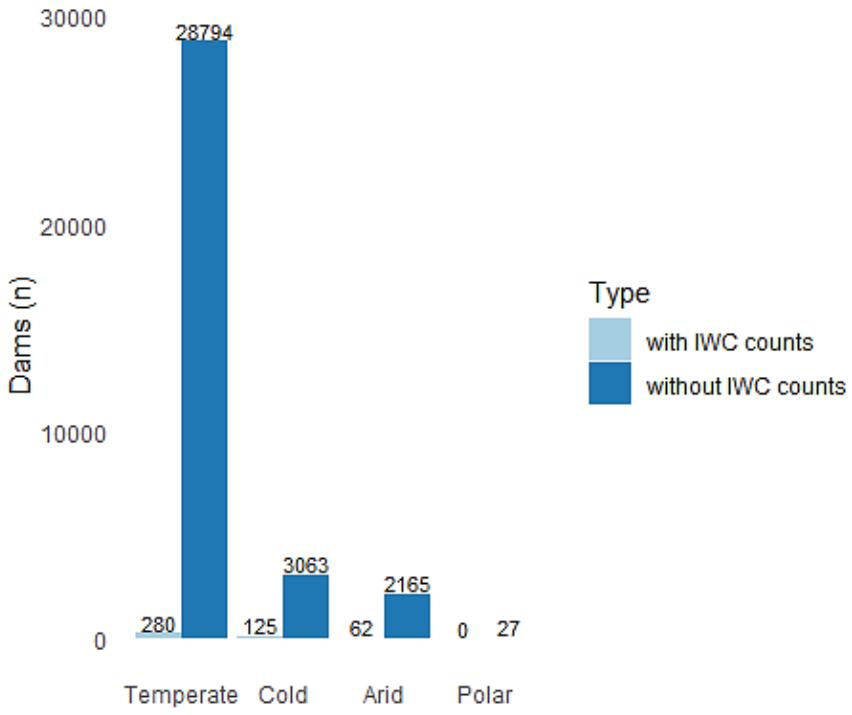


FIGURE 6 Dam numbers in each climate category for both uncounted and counted dams.

1.5. Protected areas

We found that an average of 9.5 % of dams (3290 out of 34 516 sites) were covered by protected areas (Fig. 7). IWC counts were biased towards protected dams, as 3.25% (107 sites) of the protected dams were covered by waterbird counts, and particularly towards those with an IUCN protection category ranging from I to IV (in nearly 29% of the cases). To test the association between dam protection for IUCN categories ranging from I to IV and whether or not they contained waterbird counts, we performed a Chi-Square Test of Independence. Based on the results ($X^2 = 16.77$, $p < 0.001$), there was a significant relation between dam protection and containing waterbird surveys.

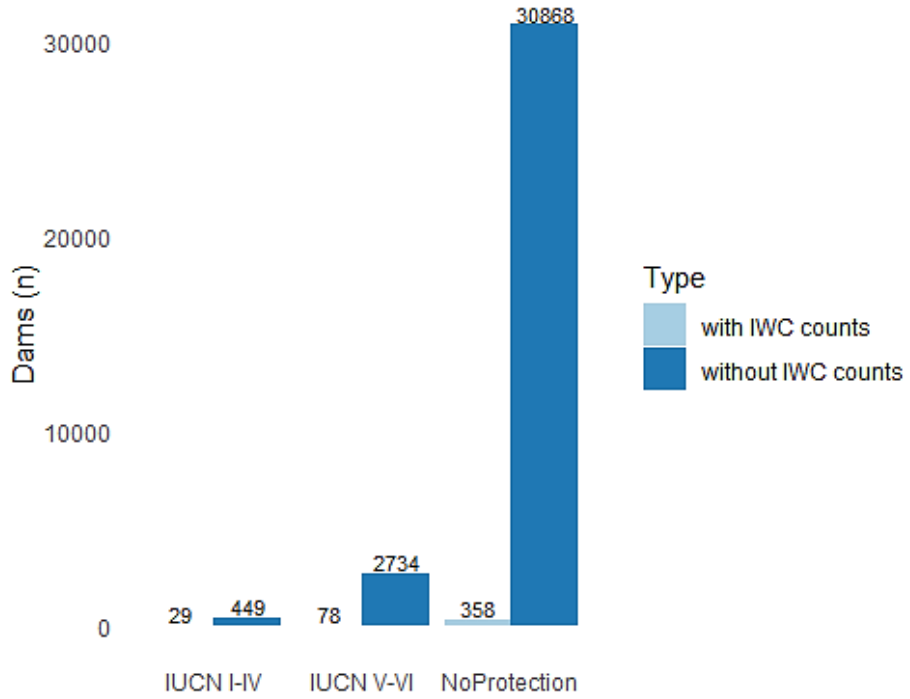


FIGURE 7 Dam numbers within both protected (IUCN categories from I to VI) and unprotected areas for counted and uncounted dams.

2. *Characterization of the sample IWC sites*

The overall comparison of IWC sites within dams versus the rest of IWC wetland sites showed that dam sites had statistically significant (Wilcoxon rank-sum test, W= 563787; $p < 0.001$) lower Human Footprint Index values (mean= 13.5; sd= 6.21) compared to the remaining wetland sites (mean= 18.8; sd= 7.86) (Fig. 8A). Moreover, dam site altitude was higher (mean= 306m; sd= 334m) than that of the IWC sites not intersecting with dams (mean= 210m; sd= 290m) (Fig. 8B). The difference in altitude was also statistically significant (Wilcoxon rank-sum test, W= 1141092; $p < 0.001$).

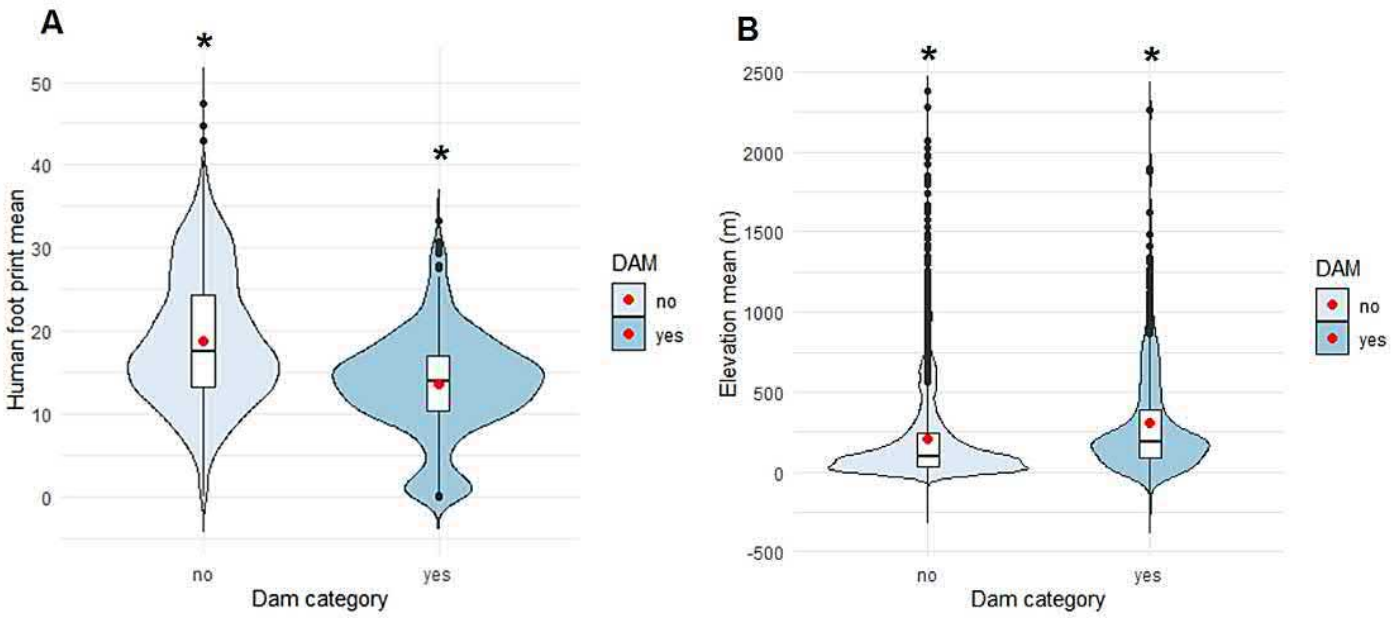


FIGURE 8 Distribution of the Human Footprint Index (HFI) (A) and mean altitude (B) across IWC sites within dams and other wetland types.

Dam sites fell within temperate climate in about half of the cases (53%), followed by cold (24.7%) and arid (22.2%) climates. Regarding the other types of IWC wetlands, the most common climate type was temperate climate (61.5%). Cold climate came next (23.1 %) followed by arid climate (15.3%), and finally polar climate (0.06%) (Fig. 9A). The difference in the distribution of climate types between IWC sites within dams and the rest of wetland sites was significant ($X^2 = 20.38$, $p < 0.001$).

We found that an average of 28.3 % IWC sites within dams were covered by protected areas, but only 3.4% had a protection level ranging from I to IV. In contrast, 40.5% of the rest of IWC wetland sites were protected, with 11.42 % of the sites with an IUCN protection status from I to IV (Fig. 9B). This difference in protection was significant ($X^2 = 44.52$, $p < 0.001$) between IWC sites within and outside dams.

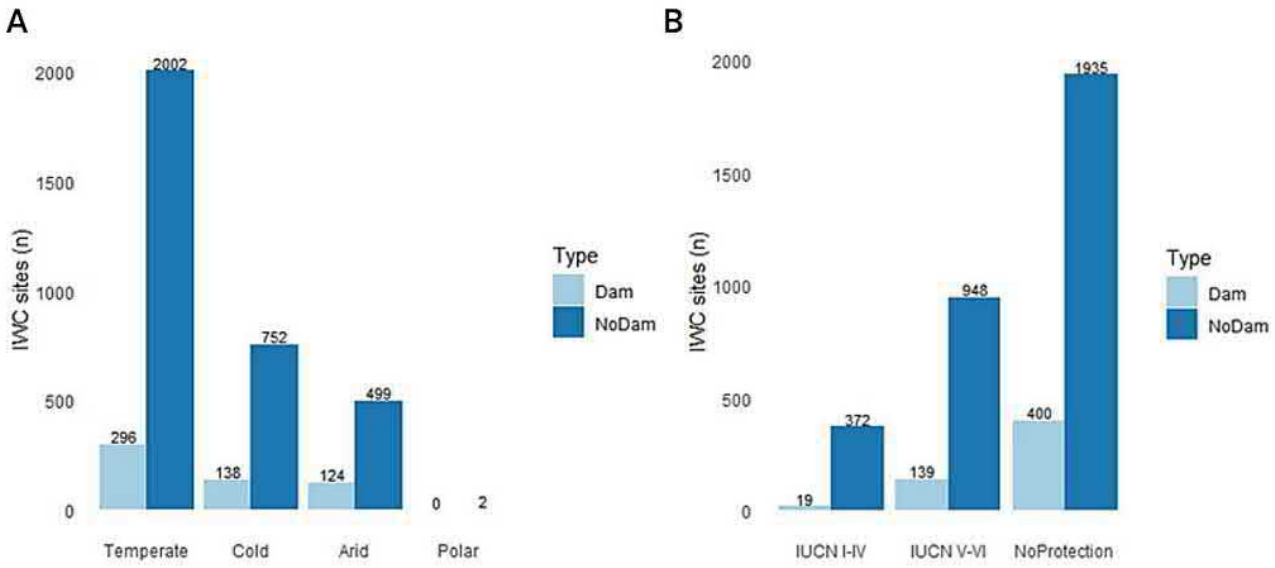


FIGURE 9 IWC site numbers in each climate category (A) and each protection status (B) for both dams and other wetland types.

3. Avian fauna composition

We recorded on average 7 632 688 waterbirds belonging to 151 species distributed in 9 orders and 18 families. The detailed list of all species, along with the total mean abundances per species in both dams and other wetland IWC sites can be found in the Appendix 1.

Overall, 123 species were recorded in dam reservoirs and 150 species in other wetland types; about 18.5 % (28 species) of the waterbird species were absent from dam reservoirs (Appendix 1). According to the BirdLife International European categorization, 19 species had a global conservation concern (SPEC1), and 39 species had an unfavourable conservation status in Europe: six concentrated in Europe (SPEC2) and 33 not concentrated in Europe (SPEC3). One of the species with a global conservation concern was found only in the other wetland types (Vanellus gregarious) (Appendix 1). The waterbird community within reservoirs was dominated by dabbling ducks (DbD) in terms of number of individuals (43.7%, 220 550 individuals) with species such as *Anas platyrhynchos*, *Anas crecca*, *Mareca Penelope*, *Spatula clypeata* occupying quite similar rank as in other wetland types (Appendix 1) except for the Ruddy Shelduck (*Tadorna ferruginea*) that occupied a higher position within dams, and by small wading birds (SWB) in terms of number of species (29.27%, 36 species).

Common coots *Fulica atra* and Mallards *Anas platyrhynchos* were omnipresent in both dam reservoirs and the rest of wetland types and occupied respectively first and second rank in both site types.

Diving species such as *Aythya ferina*, *Phalacrocorax carbo*, *Tachybaptus ruficollis* and *Podiceps nigricollis* occupied higher positions within dams compared to other wetland types. In contrast, shorebird species,

such as *Calidris alpina, Calidris minuta, Recurvirostra avosetta*, *Numenius arquata* and *Limosa limosa* principally composed were more abundant in wetland types other than dams.

4. Factors influencing waterbird species and functional group composition

Perpendicular projections of functional groups onto the HFI quantitative explanatory variable suggested that opportunistic gulls (OpG) were most associated with high pressure level sites (high HFI values), while dabbling geese (DbG), swans (DbS), small wading birds (SWB), surface fishers (SfF), dabbling ducks (DbD), diving fishers (DvF), dabbling swans (DbS), and large wading birds (LWB) were more likely to be present in sites with mid-to-high HFI values. Vegetation gleaners (VGI) and diving ducks (DvD) were closely associated with mid-to-low HFI values and were the two functional groups most associated with dam reservoirs (Fig. 10). Furthermore, the CCA showed that large wading birds (LWB) were more likely to occur in coastal wetlands, in contrast with vegetation gleaners (VGI), diving ducks (DvD) and dabbling swans (DbS).

Both the CCA terms (environmental variables) ($p<0.01$) and the CCA model ($F= 49.069$; $p<0.01$) were significant.

Functional composition variation between dams and other wetland types was confirmed by a PERMANOVA (N number of permutations = 50 000, Fmodel = 2.07, p =0.018) (see Appendix 4 for detailed PERMANOVA summary).

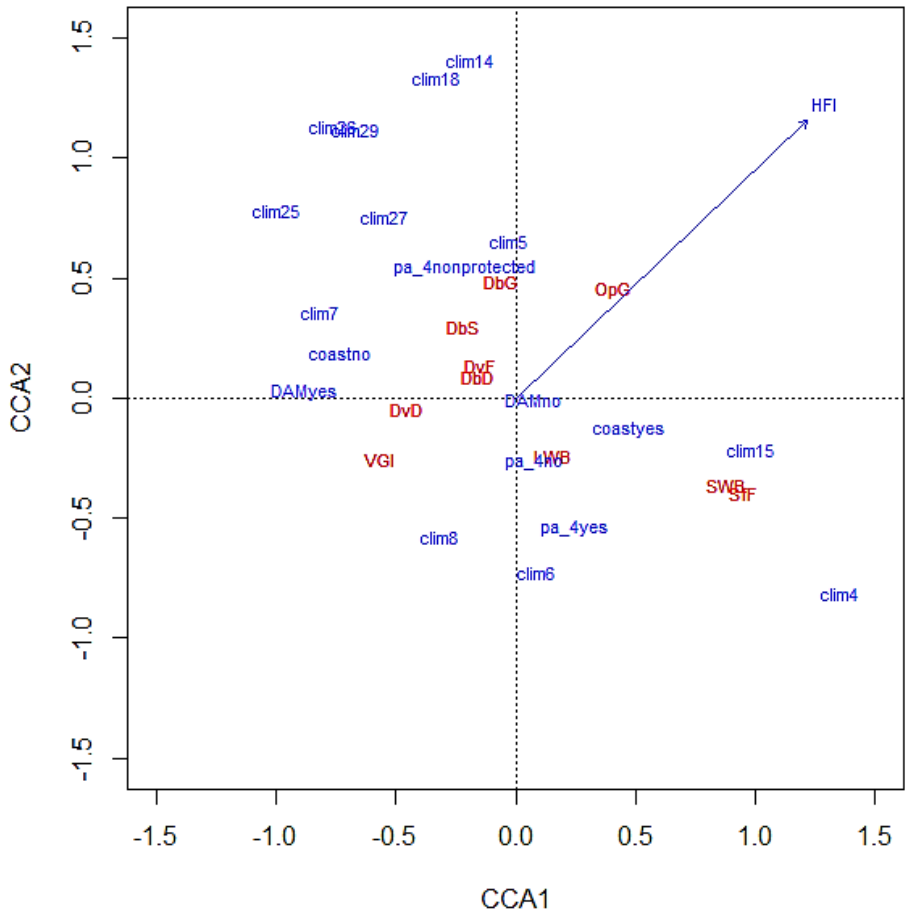


FIGURE 10 Canonical Correspondence Analysis (CCA) diagram showing functional groups and environmental gradients. This graph focuses on the ten functional group abundances, one quantitative explanatory variable (Human Footprint Index (HFI)) represented by an arrow pointing in the direction of increase, and different states of four nominal variables (DAM (yes/no), climate (see Appendix 3 for climate classes), coast(yes/no), protected areas from I to IV (yes/no/nonprotected)).

For species distribution along the environmental variables, the CCA showed a separation of the points for "DAMyes" and "DAMno" along the first axis suggesting that the two classes of sites (dams and other wetland types) differed in avian species composition (Fig. 11). Common pochards (*Aythya ferina*), Eurasian coots (*Fulica atra*), great cormorants (*Phalacrocorax carbo*), dalmatian pelicans (*Pelecanus crispus*), Whooper swans (*Cygnus cygnus*) and Bewick's swans (*Cygnus columbianus bewickii*) were most associated with dam reservoirs.

Also, from the CCA diagram we would infer that species like *Xema sabini*, *Larus marinus*, *Branta bernicla*, *Rissa tridactyla* and *Calidris maritima* were closely associated with high human pressure level landscapes (highest values of HFI), while species as *Crex crex*, *Netta rufina*, *Tadorna ferruginea*, *Anser erythropus*, *Aythya nyroca*, *Oxyura leucocephala* were very likely to be found in less human impacted landscapes (lowest HFI values).

The CCA model was validated ($F = 20.82$, $p < 0.01$) by an ANOVA as well as all the environmental variables being significantly related to the species distribution data. As with functional groups, species composition variation between dams and other wetland types was confirmed by a PERMANOVA (number of permutations = 50 000, Fmodel = 3.31, $p < 0.001$) (see Appendix 5 for detailed PERMANOVA summary).

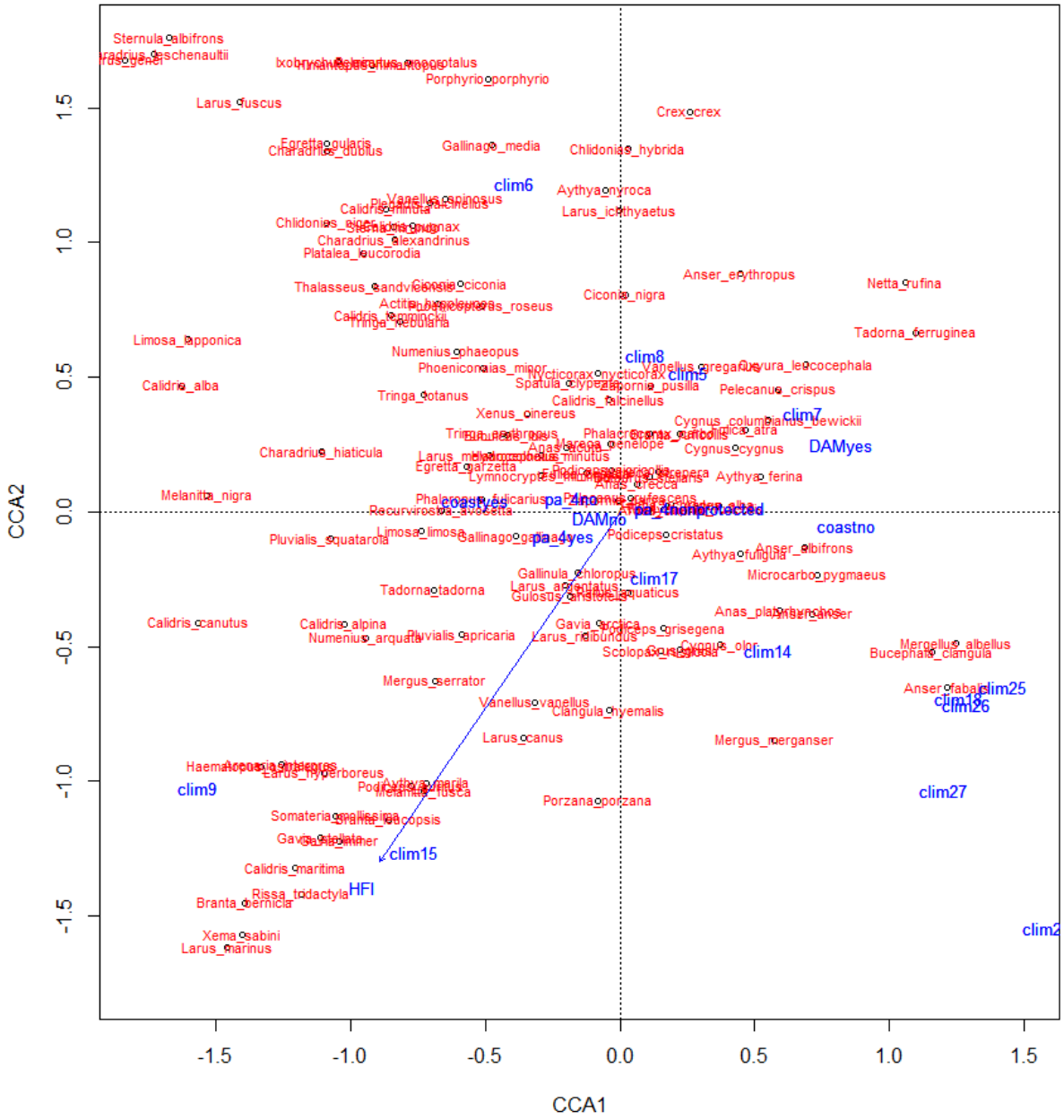


FIGURE 11 Canonical Correspondence Analysis (CCA) diagram showing both species scores and environmental gradients. This graph focuses on the 151 species-specific abundances, one quantitative explanatory variable (Human Footprint Index (HFI)) represented by an arrow pointing in the direction of increase, and different states of four nominal variables (DAM (yes/no), climate (see Appendix 3 for climate classes), coast(yes/no), protected areas from I to IV (yes/no/nonprotected)).

5. Species and functional group dam composition differences between different climate types

Waterbird composition differences between dams and the remaining wetland types were significantly climate dependent, for both species (PERMANOVA; Fmodel= 3.5, p<0.001) and functional groups (PERMANOVA; Fmodel= 3.39, p<0.001). Although groups such as dabbling ducks (DbD) and diving fishers

(DvF) were more associated to other wetlands than dam reservoirs (Fig. 12), they were more likely to be found within dams in arid climates. In contrast, dabbling geese (DbG), opportunistic gulls (OpG) and large wading birds (LWB) were more likely to be present in cold climate dam sites. As for functional groups that were most associated with dam reservoirs (Fig. 12), the vegetation gleaners' group (VGI) was related to cold climate dam reservoirs while diving ducks were more affiliated to arid dam sites.

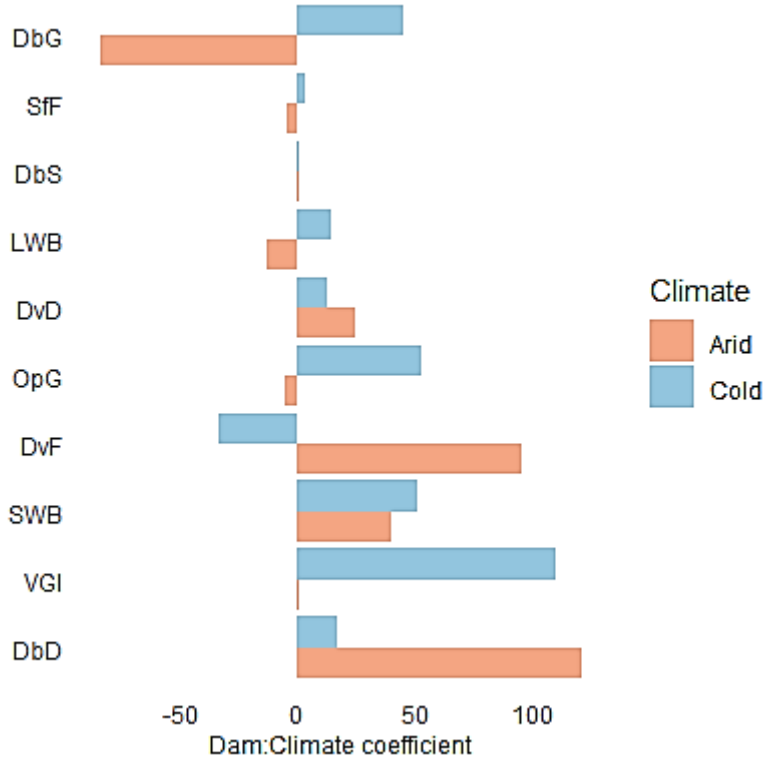


FIGURE 12 PERMANOVA coefficients of the Dam:Climate term for each functional group. Coefficients were calculated after setting the terms Dam 'yes' and Climate 'temperate' to zero.

PERMANOVA coefficients of the climate effect within dams for each species is shown in Appendix 6.

IV- Discussion

Our results show that although small dams with areas less than 1 km² were the most common type in the study area, IWC counts were biased towards larger dams. Additionally, dam reservoirs that enclosed waterbird surveys had higher Human Footprint Index (HFI) values and were situated at higher altitudes than those that didn't. Furthermore, IWC monitoring program favored protected dam sites, often with an IUCN category from I to IV. All the environmental factors [e.g., climate, Human Footprint Index, site protection, coastal and dam effect] significantly affected waterbird associations within wetland sites. We found that diving ducks were the most associated group with dam sites, whereas small wading birds, were more likely to occur in other wetland types. We also identified a significant interaction between dam and climate effects within IWC sites.

1. Result interpretation

The characteristics of surveyed dams are probably related to the site selection procedure for the census (IWC). For instance, dam sites with a higher HFI are more likely to be easily accessible to ornithologists during IWC count campaigns, as the HFI incorporates the measure of human access to quantify the degree to which humans impact the landscape (Venter et al. 2016). This may account for the observed HFI differences between dams that encompassed waterbird surveys and those that didn't. Further, large dams were the most sampled area category in the study area. This may be because site selection is based on their relative importance for waterbirds (Delany 2005), and larger wetlands tend to support more individuals but also more species than small wetlands as they usually offer higher habitat heterogeneity (Paracuellos 2006). Moreover, the highest priority for waterbird counting is given to sites with international or national designations (Delany 2005), which may explain the high proportion of protected dams that encompassed IWC counts. The fact that these dam reservoirs often fell within PAs with IUCN category ranging from I to IV is probably because the latter are subject to more restricted management objectives and prioritized use for biodiversity conservation than PAs with IUCN category from V to VI (Shafer 2015). Our results indicated that dam sites in temperate climates were poorly sampled while arid climate sites were the most sampled category, which is consistent with the regional differences in natural wetland habitat loss in the Mediterranean region. Indeed, the southeast of the Mediterranean basin, a mainly arid climate zone, has lost a larger proportion of natural wetlands compared to Southern Europe, due to a lack of wetland conservation effectiveness (Leberger et al. 2020). Thus, artificial wetlands such as dam reservoirs are more surveyed in arid areas since the latter encompass few natural wetland sites.

We found that there were significant differences in waterbird composition between dam reservoirs and the rest of wetland types as expected, probably mirroring the variation in ecological resources provided by different habitats. Diving ducks showed a close association to dam sites, whereas small wading birds, principally composed of shorebirds, were more abundant in other wetland types. These trends are probably related to water-level regulation within reservoirs. Indeed, flooding regimes play a crucial role in waterbirds distribution since water depths affect the access to foraging and roosting habitats (Murakani et al. 2015; Silva et al. 2020). As the primary purpose of dam reservoirs is to store water at the highest possible levels with minor fluctuations, exposed mudflats may become rare (Kingsford et Thomas 2004). The latter represent the main foraging habitat for shorebirds, that feed mostly on invertebrates. Gawlik (2002) showed that for wading birds that forage in the water column, deeper water can reduce their foraging efficiency since prey can escape not only horizontally, as is the case in shallow areas, but also vertically. Thus, high water levels reduce the occurrence of these species and, in contrast, benefit diving species like diving ducks (DvD) that feed in deep water. This mirrors the findings of other waterbird research within deep artificial wetlands [e.g., deep irrigation tanks in Bellio et al. (2009) and reservoirs in Li et al. (2013)]. The results of the second PERMANOVA provide evidence for the idea that under arid

climates, dam reservoirs may play a major role in hosting certain waterbird functional groups, even when they are originally associated to other wetland types. This finding matches the regional wetland loss context within the Mediterranean basin, where southeastern parts had undergone an accelerated degradation of natural wetlands (Leberger et al. 2020).

2. Conservation implications

On dam reservoirs, dabbling ducks were the most numerous in terms of number of individuals. This trend was quite similar within the rest of wetlands except for the Ruddy Shelduck *Tadorna ferruginea* that occupied a higher rank within dams. It is reported that this species have isolated and declining populations of less than 10 000 individuals in the West Mediterranean (Green et al. 2002) and an unfavourable conservation status in Europe (BirdLife International 2004). Besides the Ruddy Shelduck, dam reservoirs hosted high numbers of threatened waterbird species of high conservation value, such as the White-headed Duck *Oxyura leucocephala* (SPEC1), the Northern Lapwing *Vanellus vanellus* (SPEC1) and the common Pochard *Aythya ferina* (SPEC1). It should be noted that out of the nineteen species with a global conservation concern (SPEC1), only one of them, the sociable Lapwing *Vanellus gregarius* was found exclusively in wetlands other than dams. This implies that dam reservoirs seem capable of providing appropriate habitat to species of international conservation importance and thus can be of considerable conservation value. Similar results were found elsewhere [e.g., the White-headed Duck found in the Akhna Dam in Cyprus (Giosa et al. 2018) and in hill dam reservoirs in Tunisia (Afdhal et al. 2013)].

As highlighted before, small dams were the most common type in the study area but the least sampled category in the IWC monitoring program. Additional waterbird surveys should be organized including these reservoirs, especially as previous studies have suggested that a set of small wetlands could support the same or even greater waterbird richness as one large wetland with equal area (Scheffer et al. 2006).

3. Framework significance, limitations, and perspectives

Other studies were often limited to evaluating waterbird community composition within dams at a restrained spatial scale [e.g., Morocco in Green et al. (2002), Northern Tunisia in Hamdi (2014) and Afdhal et al. (2013) and finally Cyprus in Giosa et al. (2018)]. Additionally, their study periods were also restrained. This makes our results more significant since we provide information on the size of waterbird populations wintering in dams in 22 countries for a 25-year period. Additional studies should be programmed including Algeria, Spain, and Libya to assess the conservation value of dam reservoirs at the Mediterranean scale. Especially as Algeria and Libya fall within arid climates and that Spain encompasses the highest number of large dams in the Mediterranean region (MedECC 2020). This is even more important given the major resurgence in dam construction in the basin, particularly in countries with emerging economies such as the Balkans (MWO2, 2018).

However, our results must be interpreted with caution for methodological reasons. First, although not all wetlands were surveyed every year due to a lack of financial or human support, we chose to compute mean species abundances over the count period for each IWC site. We adopted this approach, rather than considering only sites that were surveyed at least twice during the study period as documented in (Gaget et al. 2020), to ensure that our sample size was large enough to be representative and to use statistical approaches, which may misestimate species abundances within the least surveyed sites.

Second, the waterbird census GIS data does not contain polygon features for most countries, and thus did not allow us to include wetland size in our analysis. Several studies have documented that wetland size influences abundance and richness of waterbirds (Froneman et al. 2001; Ma et al. 2004; Sánchez-Zapata et al. 2005; Ma et al. 2010; Afdhal, Charfi, et Moali 2013). Hence the importance of additional studies considering biophysical features of wetlands, including area, that would also allow to incorporate these characteristics into dam construction processes and thus ensure that reservoirs are designed and managed in a way that limits their impact on waterbirds populations.

Finally, our study focuses only on the wintering period, so further investigation of the distribution and the status of waterbird communities within dams in both breeding and migration seasons is very much needed. This is even more crucial as migrating waterbirds require suitable conditions for resting and foraging to fuel their migration and thus foster a successful breeding season (Barragán Severo et al. 2002).

4. Conclusions

To conclude, given the rapid wetland loss that the Mediterranean region is undergoing (MWO2, 2018), the conservation value of man-made wetlands, notably dam reservoirs, become increasingly apparent for several species, particularly under arid climates. Therefore, conservation actions should prioritize the protection of biodiverse reservoirs in regions where natural wetlands are the most unavailable.

It is here suggested that the conservation value of reservoirs should not be discounted in the Mediterranean context. However, it is important to remind that over-emphasizing the role of dam reservoirs in hosting diverse waterbird communities may inappropriately encourage landowners to convert natural wetlands, thus accelerating the loss of natural habitats. Although it is judicious that conservation actions should prioritize natural wetlands, this can be complemented by the protection of biodiverse dam reservoirs.

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ABSTRACT

Wetlands are considered as one of the most threatened habitats worldwide. In the Mediterranean basin, they are of major concern, facing a 45% loss since 1970. In this context, artificial wetlands such as dam reservoirs are playing an important role in maintaining avian diversity. Waterbird communities using reservoirs are generally less diverse than those using natural wetlands. However, the conservation value of these habitats needs further exploration given the current major resurgence in dam construction within the Mediterranean region. Here, we evaluated wintering waterbird community composition within dam reservoirs for 22 Mediterranean countries between 1993 and 2017. We characterized dam reservoirs that encompassed waterbird surveys, and tested the dissimilarities in both species and functional group composition between dam sites and other wetland types. The waterbird community within reservoirs was dominated by dabbling ducks in terms of number of individuals. We found that there were significant differences in both functional and species composition between dams and other wetland types. Diving ducks showed a close association to dam sites, whereas small wading birds were more abundant in other wetland types. Our results also highlight the importance of dam reservoirs for waterbird species of high conservation value such as *Oxyura leucocephala* and *Vanellus vanellus*, and provide evidence that under arid climates, dam reservoirs may play a major role in hosting waterbird communities, even when they are originally associated to other wetland types. We conclude, from our results, that conservation actions should prioritize the protection of biodiverse reservoirs in regions where natural wetlands are the most unavailable. This study should be further extended to include Algeria, Spain, and Libya to assess the conservation value of dam reservoirs at the Mediterranean scale.

Keywords: Artificial wetlands, conservation, dam reservoirs, Mediterranean basin, waterbirds.

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