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Isadora Christel Jiménez García
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“Herzog dijo que hay más de un Anapurna en la vida
de cada hombre y no siempre se alcanza la cumbre…
pero si el esfuerzo nos permite vislumbrar algo de aquello
que hay más allá del azul infinito, ya vale la pena.
Al menos ayuda a vivir.”

(1980, carta de Enric Benavente i Mata a mi abuelo, Antonio Mega)

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ACKNOWLEDGEMENTS

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Estas líneas abren la tesis pero son las últimas que escribo. Es el final del camino. Un camino largo, no siempre fácil, pero desde luego lleno de gente que me ha ayudado, apoyado, animado y que de una forma u otra ha estado a mi lado mientras este proyecto iba madurando.

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Mis primeros pasos por el departamento de Biología Animal empezaron mucho antes de acabar la carrera, era 2001 y después de hablar con Domingo acabé haciendo campañas de campo capturando codornices y descuartizando tejones. Aún hoy soy hipersensible al canto de *Coturnix coturnix*. **Domingo**, eres un magnífico profesor de campo. El tiempo que colaboré contigo, Manel y con todos tus entonces doctorandos fue lo que me hizo decidirme a dar el paso hacia la ciencia, así que para ser justos, ese es el principio del camino.

Xavier, encara recordo entrar al teu despatx per parlar del treball de Zoogeografia, l'últim treball de la carrera… Ha plogut molt des de la idea per aquell treball fins arribar a aquesta tesi, el tema ha donat voltes però crec que podem sentir-nos orgullosos. Gràcies per la oportunitat d'arribar fins aquí, per entendre i donar suport a la noia pràctica que sóc i per confiar en mi tot i els daltabaixos inevitables en tants anys.

Els principis van ser solitaris, lluny de la universitat i amb trasllats de despatx en despatx. Per sort els vaig poder compartir amb tu, **Albert**. També recordo el nostre primer té/cervesa a la UPC… No sempre ha estat fàcil treballar plegats. Tots dos som d'idees clares i tossuts per defensar-les, però crec que hem superat prou bé les nostres diferències. Em quedo amb les reunions al bar que han solucionat més d'un problema metodològic, d'anàlisi o de com enfocar les nostres tesis. I bé, també amb unes quantes sessions de radar al Delta en companyia de Xesco.

Xescuu! De ti me quedo con los intentos por iniciarme en el mundo de la ornitología, pero sobre todo, con tu buena música para pasar las horas delante de un monitor en medio de la nada.

El traslado al departamento fue un cambio agradable. La tesis pasó de ser una lucha solitaria a una montaña rusa compartida con otros sufridos becarios. En compañía, todo se hace más llevadero sobre todo si va servida por **Jose Luis** y **Miguel**, sois mis *biobarmans* preferidos. En esta montaña rusa y desfile de compañeros de bar, pasillos y despachos hay muchos nombres. **Eloy**, gracias por entender a la perfección esos días en los que voy dando saltos por los pasillos y por darme ese abrazo siempre que lo he necesitado. **Gemmmma**, gràcies per donar geni i figura als dies grisos de la facultat. **Cotín**, por descubrir mi *wild side*. **Blanca**, por ser el eslabón que une a todos los departamentos. **Mari Carmen**, por dar color a la facultad con tus viajes y experiencias. **Alberto**, por ser el perfecto asesor estratega. **Olatz**, por ese trabajo de campo en bikini en el principio de los principios. **Edu**, por recordarme que la ciencia al más puro estilo "Konrad Lorenz" aún es posible, por hacerme sentir no tan R-freaky y por todas esas charlas entre mis cafés y tus cigarros. Pero es injusto que haya tan poco espacio, porque son muchos nombres y de todos podría decir algo: **Manolo**, **Debs**, **Victor**, **Irene**, **Mario**, los "(no tan) nuevos": **Jose Manuel, Jaime**, **Fav**, **T**, **Nicole, Urtzi**… y los que ya dejasteis el nido: **Oriol**, **Laura "arpella"**, **Inés**, **Luigi**, **Rubén, Josep Lluis, Eva.** Dejo muchos nombres por poner… no me lo tengáis en cuenta porque hoy me acuerdo de **todos** vosotros.

La tesis hizo un salto cualitativo cuando una llamada me informó de que, cuando ya había perdido toda esperanza de obtenerla, me daban la ansiada beca predoctoral. Con la beca también llegó mi primer co-director, **Vieites**. Nos hemos visto muy poco (aún te llamo por tu apellido!) pero ha sido tiempo altamente productivo. Ese retiro en la sierra madrileña fue una experiencia genial y siento que el espacio-tiempo no se haya alineado para poder interaccionar más. Aún con todo, me alegro de tenerte como "codire".

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Si hay algo a destacar, es el cambio de inflexión que supuso para mí ir a Noruega. Norway. *Norgue*. In Trondheim I found my third co-supervisor and the focus of the thesis that I had been unconsciously looking for. **Grégoire**, merci de ton soutien. The first day we met at the airport I already had the feeling that we were going to get on well. I love your scientific stubbornness, your freak side and your quality as supervisor, mentor, advisor… You helped me to jump without doubts to the world of R programming, and gave me such a quantity of useful tips and comments for my future as a researcher that I had professionally changed by the time I went back to Barcelona. You have the makings of a wonderful supervisor. Take more students! *Tusen takk* also to **Tycho, Inga, Duncan, Signe, Camilla** and **all and each of the NINA colleagues** that opened my mind and gave me so much to bring back home. And well, the experience was wonderful, but it was also possible thanks to you, **Vidar**. With you at home I didn't feel the Norwegian cold. You adopted the *Spansk* roommate and immersed me completely into Norway with our afternoons of talking, watching football, movies and partying. My second stay in Norway was a really short one, but those few weeks at the IMR in Tromsø (almost 70ºN) were also a very intense experience full of new colleagues for science: **Benjamin, Alf Harbitz, Ulf** and leisure: **Maria,**

Marina, Atal, Keka, Rune, Trond, Virginie, **Roland**, **Bas**… thank you for everything and more.

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Ya hacia el final descubrí toda una vocación, la comunicación científica. 2011 fue un año intenso, pero junto a mis compis de Máster pasó volando. **MCC16 forever!** No pongo vuestros nombres porque sois muchos, pero de verdad que esta tesis llega a buen puerto gracias a esa transfusión de energía, ganas y motivación que compartimos de 7 a 10 de la noche cada día.

Y como no… padrins, tiets, primos, abuelos y padres, propios y políticos: **Familia**. Gracias por acompañarme aún sin acabar de entender demasiado qué hacía o por cuánto tiempo iba a “seguir estudiando”. Está claro que por más que lo he intentado no he conseguido convenceros de que esto ES UN TRABAJO así que: Abuelos, por fin la nieta “ha dejado de estudiar”!

Como ya he dicho, para mí una tesis es una montaña rusa. Tiene subidas y bajadas. Mis estancias en Noruega y el máster en comunicación científica están en las cimas, pero también ha habido unas cuantas bajadas vertiginosas y en esos momentos he tenido la fortuna de contar con tres refugios, remansos de paz, fuentes de consejos y ánimos. Tres lugares en los que tal cual cruzar la puerta se me ha permitido descargar de mis hombros el peso del doctorado, los miedos y todas las dudas. Uno está en Palamós-Gracia-Centelles... allá dónde sea que **Laura** prepare una infusión y **Joan** improvise una cena. Vuestra casa siempre ha sido un refugio de calma y consejos en los momentos más críticos. Otro está en Amsterdam. **Alicia**, gracias por todos estos años de amistad y vuelos de ida y vuelta, que no es poco. Pero independientemente del lugar, los brazos de **Ivan** han sido mi mejor refugio.

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HOW IT ALL BEGAN

"What do you think? Will it have an impact?"
"Oooh, so you moved to the dark side… you will say yes to the wind farm, won't you?"

Those were the two most asked questions at the beginning of my research experience. In August 2004 Dr. Xavier Ferrer was commissioned the Environmental Impact Assessment for an offshore wind farm. A project in front of the Ebro Delta area. The developer company, Capital Energy Offshore, paid for an exhaustive EIA and Albert Cama and I suddenly found ourselves in the middle of a huge, potentially conflictive project. No complaints! In some countries, paid science is a luxury.

People from Capital Energy, my family, friends... all used to ask me the first question. My biologist colleagues would look at me with terrified faces and go for the second comment/question. My answers:

"Yes, there will be an impact; we don't need the study to say this"

"No, it is not my job to decide that"

This situation was the seed of this thesis because there was not much scientific literature on the topic. After some months of bibliography search, all I could find was grey literature from governments, research institutions and the offshore wind industry. Some reports were helpful as they gave recommendations on survey methodology but all their conclusions seemed a sequence of "too many" descriptive distribution maps summarized at the end with a few paragraphs according to the previous ornithological knowledge of the authors.

As I see it, in this topic, our role as scientists is to evaluate, in the most impartial and objective way, the impacts; and more importantly to do our best for quantifying them. With this information, we have to inform decision-makers in the most clear synthetic way. A French supervisor of mine would argue that not even scientists can reach real impartiality and objectivity, but I am satisfied if we make a sincere effort to reach them.

During these years, I have been somewhere in-between ornithologists, ecologists, conservationists, managers and business people. It is not easy when you do not fit in a particular label, but at least it gives you a different point of view. After 8 years working in the University, I have learnt a few things about myself: 1) I am definitively not a passionate ornithologist (although now I can even follow their conversations); 2) I definitively like methods and programming; and 3) I love visual communication of concepts because I have a taste for simplicity. This thesis is the result of this.

I wanted to bring some integrative tools to summarize results in the fewer maps as possible. I wanted to transcend plain descriptive distribution maps, “eradicate” them or at least move them to the appendix of any Impact Assessment. There is a huge part of fruitless work that has no space in this dissertation and there is still a great part of work to be done. I know. But I got to the end of the fourth year of my University PhD grant and I took an unpaid extra half year. Luckily for a scientist, what is still to be done is just an opportunity for the future.

General introduction

INTRODUCTION

“Offshore wind energy development and seabirds’ conservation: A management challenge”

The world’s growing energy demand and climate change are two of the great challenges of this century. A trade-off between climate change policies and competitiveness is needed to find an economically viable low carbon future. In this context, the European Commission defined the ‘Energy roadmap 2050’ that explores the possibilities to achieve a low-carbon economy that at the same time ensures a competitive, sustainable and secure energy supply (EC, 2011a). The European Union is committed to reducing greenhouse gas emissions up to 80-95% below 1990 levels by 2050 (EC, 2011b). It is impossible to forecast Europe’s long-term evolution but some of the possible low-carbon scenarios are (i) a highly energy efficient system, (ii) a system with a diversified supply of technologies including carbon capture and storage facilities and/or nuclear power, and (iii) an scenario with a strong support to renewable energy sources. All the predictions for these decarbonized Europe scenarios show that electricity will have to play a greater role than fossil fuels and the share of renewable energy sources will rise substantially achieving at least the 55% of the gross final energy consumption in 2050 up to 64% or 97% depending on the scenario (EC, 2011b). One of the policy measures to achieve this goal is the Renewables Directive, which sets a target of 20% of energy consumption to come from renewable sources in 2020.

In Europe, renewable energy sources represent the 18% of all the energy production (Eurostat, 2009; Fig. 1a). Within Renewable electricity production, hydropower is the main source (54.5%) followed by wind power (22.5%) (Observ’ER, 2011; Fig. 1b). By 2050, wind power is expected to provide more electricity than any other technology (EC, 2011b) and hence the potential contribution of the marine environment for offshore wind energy development has received high attention in the last decades.

OFFSHORE WIND ENERGY

The first offshore wind farm was installed in Denmark in 1991. Since then, the sector had a rapid expansion (Fig. 2), particularly in the North of Europe. So far, Europe has become the world leader in offshore wind power with a total of 1371 offshore turbines spread across 53 wind farms in 10 countries by the end of 2011 (EWEA, 2012). The UK is

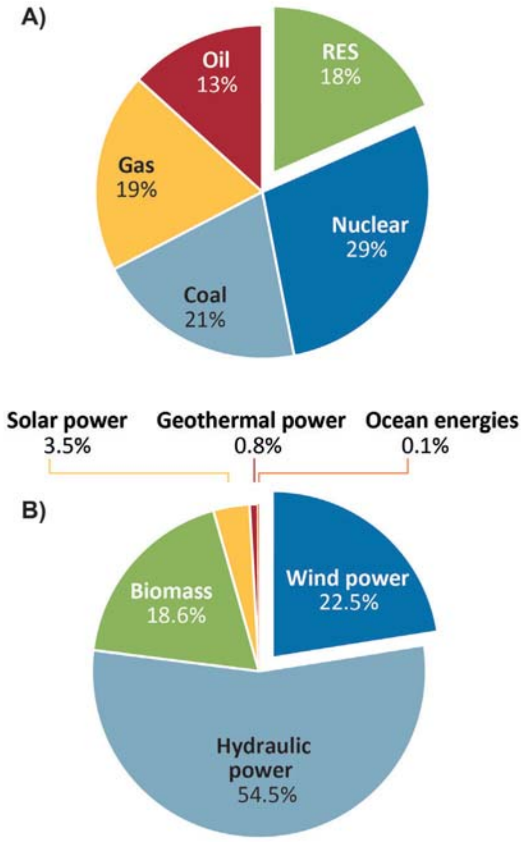


Fig.1 a) *EU energy production by source in 2009 (Eurostat, 2009) RES=Renewable Energy Sources.* **b)** *Share of each resource in Renewable electricity generation in 2010 (Observ'ER, 2011).*

Box 1 *List of commonly used abbreviations.*

ABBREVIATIONS

OWF: *Offshore Wind Farm*
SEA: *Strategic Environmental Assessment*
EIA: *Environmental Impact Assessment*

the country with the largest installed offshore wind capacity, followed by Denmark, Netherlands and Germany (Table 1). Interest in offshore wind energy is spreading beyond Europe. China, Japan, South Corea, USA and Israel have companies actively developing offshore wind turbines, although only China has three operational offshore wind farms.

Most of the installed turbines have foundation structures. Floating models are being developed, and Norway and Portugal are the first countries that have a full-scale floating turbine installed. As the technology matures, offshore wind farms are expected to grow in size but also to be deployed further from the coast and in deeper waters, particularly if floating technology is further tested and its economic viability demonstrated. Current projects under construction have an average depth of 25 m and a distance to the shore of 33 km (EWEA, 2012). This is possible because many of the actual OWF have been built in the North Sea that has a large part that lies on the European continental shelf (Fig. 3). This provides relatively large flat and shallow regions suitable for development (Henderson et al., 2003). In comparison with Northern Europe, the West coast of France, the Iberian Peninsula and the Mediterranean Sea remain a challenge for OWF development. Although there are planned projects for these areas, the available turbines and foundation methods would require the construction of the wind farms much closer to the shore with a consequent increment of the conflicts to find optimal locations in terms of social acceptance, environmental impacts, conflicts of interest and national marine spatial planning. All these factors, together with a lack of funding, are slowing the offshore development of wind energy in West and South Europe.

Indeed, the offshore wind energy is by no means free of conflicts. At a global scale, the

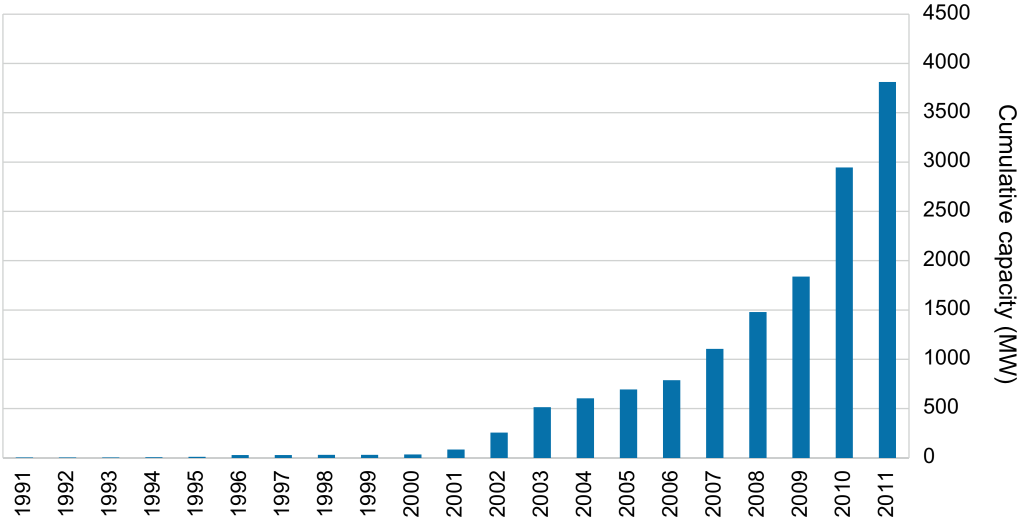


Fig.2 *Cumulative offshore wind installations (MW) (EWEA, 2012).*

Table 1 *European Operational Offshore wind farms by country (EWEA, 2012). UK: United Kingdom; DK: Denmark; NL: Netherlands; DE: Germany; BE: Belgium; SE: Sweden; FI: Finland; IE: Ireland; NO: Norway; PT: Portugal.*

Country	UK	DK	NL	DE	BE	SE	FI	IE	NO	PT	Total
No. of farms	18	13	4	6	2	5	2	1	1	1	**53**
No. of turbines	636	401	128	52	61	75	9	7	1	1	**1371**
Capacity installed (MW)	2094	857	247	200	195	164	26	25	2	2	**3813**

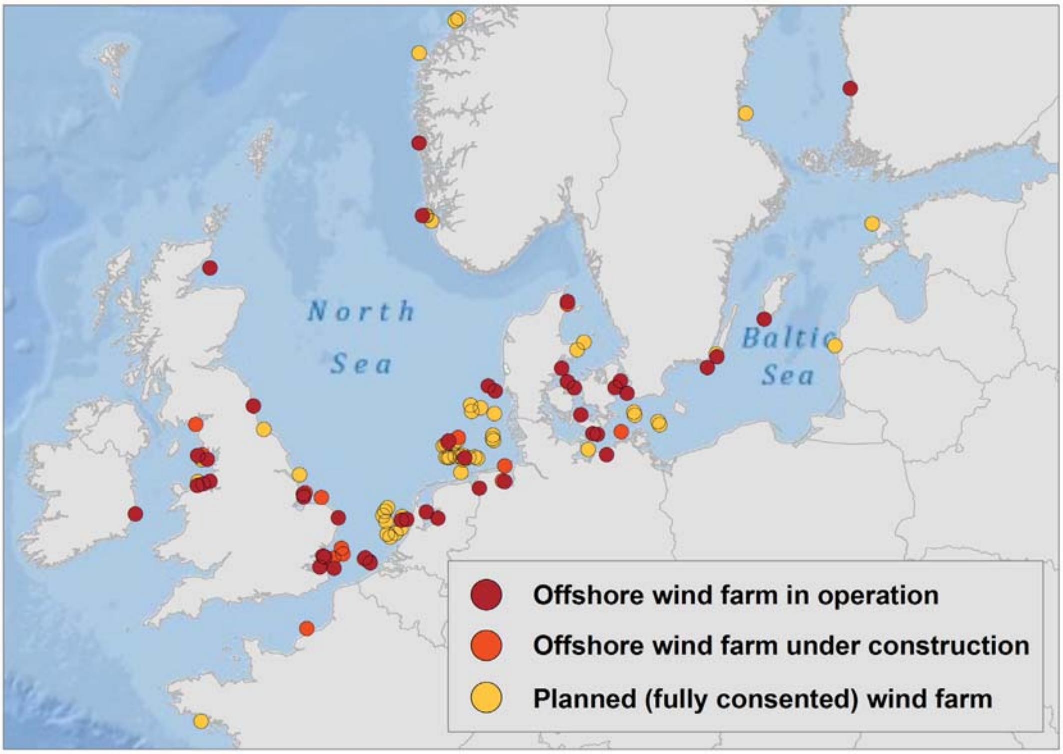


Fig.3 *Operational and planned offshore wind farms in Europe (EWEA, 2011).*

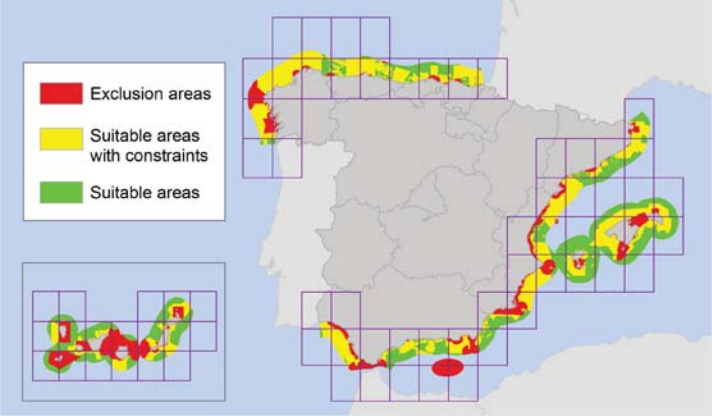
shift to renewable energies is widely accepted as a necessary step to mitigate the effects of anthropogenically induced climate change (King, 2004; Rosenzweig et al., 2008). At the local scale, however, the environmental impacts of wind energy development must be carefully considered (Gill, 2005). In the field of marine management, there is a growing concern on the development of offshore wind energy and its potential impacts on the marine ecosystem. Some of the aspects that are being studied are the disturbance of the seabed and fauna during OWF construction and operation (Whitehouse et al., 2010; Burkhard et al., 2011), the impacts on fish larvae (Perrow et al., 2011), the unknown effects of underwater noise on fish life and sea mammals (Madsen et al., 2006; Bailey et al., 2010) and the effects at population-level of collisions of birds with turbines (Fox et al., 2006; Desholm, 2009) and disturbance (Drewitt and Langston, 2006; Masden, Haydon, et al., 2010).

Box 2 *Summary information of PTTs performance*

OFFSHORE WIND ENERGY IN SPAIN

Spain has no operational offshore wind farms so far. Since the beginning of the offshore wind energy expansion in Northern Europe, different developer companies showed their interest on constructing offshore wind farms in the Spanish coasts. Despite the early private sector initiatives to promote its development, the Spanish government took the first legislative step forward in 2007. That year the Real Decreto 1028/2007 was published setting the compulsory administrative procedure that developers should follow to have the concession to construct an offshore wind farm in the Spanish coasts.

As part of the necessary procedure a Strategic Environmental Assessment (SEA) for the Spanish coastline was commissioned. This study was published in 2009 (MARM and MITYC, 2009) and included the definitive zonation map for offshore wind development areas. This map divided the Spanish coasts in 72 marine eolian areas (defined by one decimal degree squares). Within each area, the 24 first nautical miles were assessed according to multiple criteria and classified as suitable areas (in green), suitable areas with constraints (in yellow) and exclusion areas (in red).



The administrative concessions process is long and complex and has suffered several delays. To the date, the start of the application process is on hold, hence there is no official number of planned wind farms in Spain.

ENVIRONMENTAL ASSESSMENT

The European Union has a regulatory framework (Directive 2001/42/EC) to standardize the evaluation and monitoring of human activities in the ecosystem and to guarantee a rational development of such activities including environmental considerations. On a large scale, countries must develop a Strategic Environmental Assessment (SEA) to plan their offshore wind farms network minimizing their ecological impact on the coastal environment. At a local scale, each wind farm project requires an Environmental Impact Assessment (EIA) of the possible negative impacts of the proposed project in the marine environment.

The EIA concept was first introduced in a European Directive in 1985 (Directive 85/337/EEC) but it was restricted to certain types of projects. Years later, the need to deal with environmentally damaging decisions at national levels developed into the Strategic Environmental Impact Assessment that was finally included in a European Directive in 2001. Although all countries of the EU are implementing SEAs since 2004, EIAs have a longer tradition and clearer implementation procedures. This is also reflected in the environmental assessment of offshore wind energy.

For many years, the only available information on offshore wind farms assessments were reports focused on how to perform EIAs of particular projects. The Danish experience with the first wind farms was extensively reported by the National Environmental Research Institute (NERI) and their aerial surveying methodology has become a standard for many EIAs (Noer et al., 2000). Later, the COWRIE (Collaborative Offshore Wind Research Into the Environment) from UK commissioned a report to standardize the seabird surveys techniques for EIA of offshore wind farms (Camphuysen et al., 2004). In the last years, as the sector has grown, more reports have been published as well as research papers on the assessment of environment-OWF interaction of particular wind farms (*e.g.* Desholm and Kahlert, 2005; Perrow et al., 2011; Skeate et al., 2012) as well as reviews and general papers regarding SEAs (Elliott, 2002; Fox et al., 2006; Punt et al., 2009; Masden, Fox, et al., 2010).

SEABIRDS AS INDICATORS

Marine Ecosystems are highly biodiverse and ecologically complex. While ecological studies focus on this complexity, applied ecology requires methods that synthesize this complexity in order to take actions that may have economic consequences (Piatt and Sydeman, 2007). Such is the case of using indicator species to simplify the monitoring and management processes for EIAs and SEAs. Marine top predators are a key component of marine ecosystem management (Boyd et al., 2006) and within top predators, seabirds have become widespread indicators to evaluate potential effects of human activities at sea as well as ecosystem health (Cairns, 1987; Nettleship and Duffy, 1993; Mallory et al., 2006).

Seabirds offer many advantages compared to other species. Considering an environment where most species are under water, seabirds are conspicuous animals, they are easily surveyed during their movements and in resting areas; and some species are easy to capture allowing individual tracking and demographic studies (Piatt and Sydeman, 2007). Moreover, most seabirds have specific legal protection frameworks (*e.g.* Birds directive and Habitats directive in Europe) and are flagship species for the public (Fox et al., 2006) which is reflected in the abundance of comprehensive long-term studies of their distribution at sea and population trends.

Because of all this, the distribution and abundance of seabirds are usually provided as key information to support the establishment of marine protected areas (Garthe et al., 2011; Arcos et al., 2012), to implement fisheries' management measures (Boyd et al., 2006), to monitor the impact of oil and gas platforms at sea (Wiese et al., 2001), or to assess the impact of environmental disasters such as oil spills (Bretagnolle et al., 2004; Moreno, 2010). Thus, seabirds are suitable indicators of the marine environment, and have become one of the keystones of the decision-making process for the selection of optimal areas for national offshore wind development and the impact assessment of particular OWF projects.

Potential impacts on seabirds

At the time of selecting development areas, or when the location for a project is settled, we can differentiate the effect of OWF on two types of seabirds: migrant species that may encounter the wind farms in their migratory routes and breeding and wintering species with wind farms in their foraging grounds. Both types of seabirds are susceptible to multiple anthropogenic impacts (*e.g.*Anderson et al., 2003; Hüppop et al., 2006; Louzao et al., 2006), but the potential impacts of offshore wind farms on seabird communities can be classified in three types; (i) direct mortality through collision, (ii) modification of their physical habitat and (iii) avoidance due to disturbance and barrier effects.

Collision risk

Birds flying within a wind farm area are clearly at some risk of colliding with the blades, the stationary structure or being caught and injured in the pressure vortices created by the rotor blades (Fox et al., 2006). Collision risk depends on a range of factors related to bird species (manoeuvrability, wing span, etc.), behaviour (*e.g.* nocturnal activity), presence in large numbers and weather conditions reducing visibility. Collision mortality is the most important hazard since direct mortality can potentially have rapid consequences at population levels. Nevertheless, there is still limited information on the actual numbers of bird collisions with offshore wind farms, largely as a consequence of the technical difficulties to detect these collisions at sea (Drewitt and Langston, 2006).

Gradually, more remote technologies are being included in the study of bird-turbine collisions at offshore wind farms. One of the most extended tools are S-band Radars although they cannot quantify collisions directly and depending on the study cannot provide species specific information (Christensen et al., 2004; Desholm et al., 2006). Yet, radars are a useful tool to implement collision models (Desholm and Kahlert, 2005; Chamberlain et al., 2006). Thermal Animal Detection Systems (TADS) are also an alternative to gather information on actual collision rates. This infrared based technology, however, has been seldom applied and there are few published studies on its performance (see Desholm et al., 2006 for a review).

Habitat change

This impact comprises the loss of habitat that would result from the presence of the turbine bases, grid connection cabling and any other associated construction. The scale of habitat loss is not generally perceived as a major concern whenever this is not produced in areas of high biodiversity or ecological importance (BirdLife International, 2003).

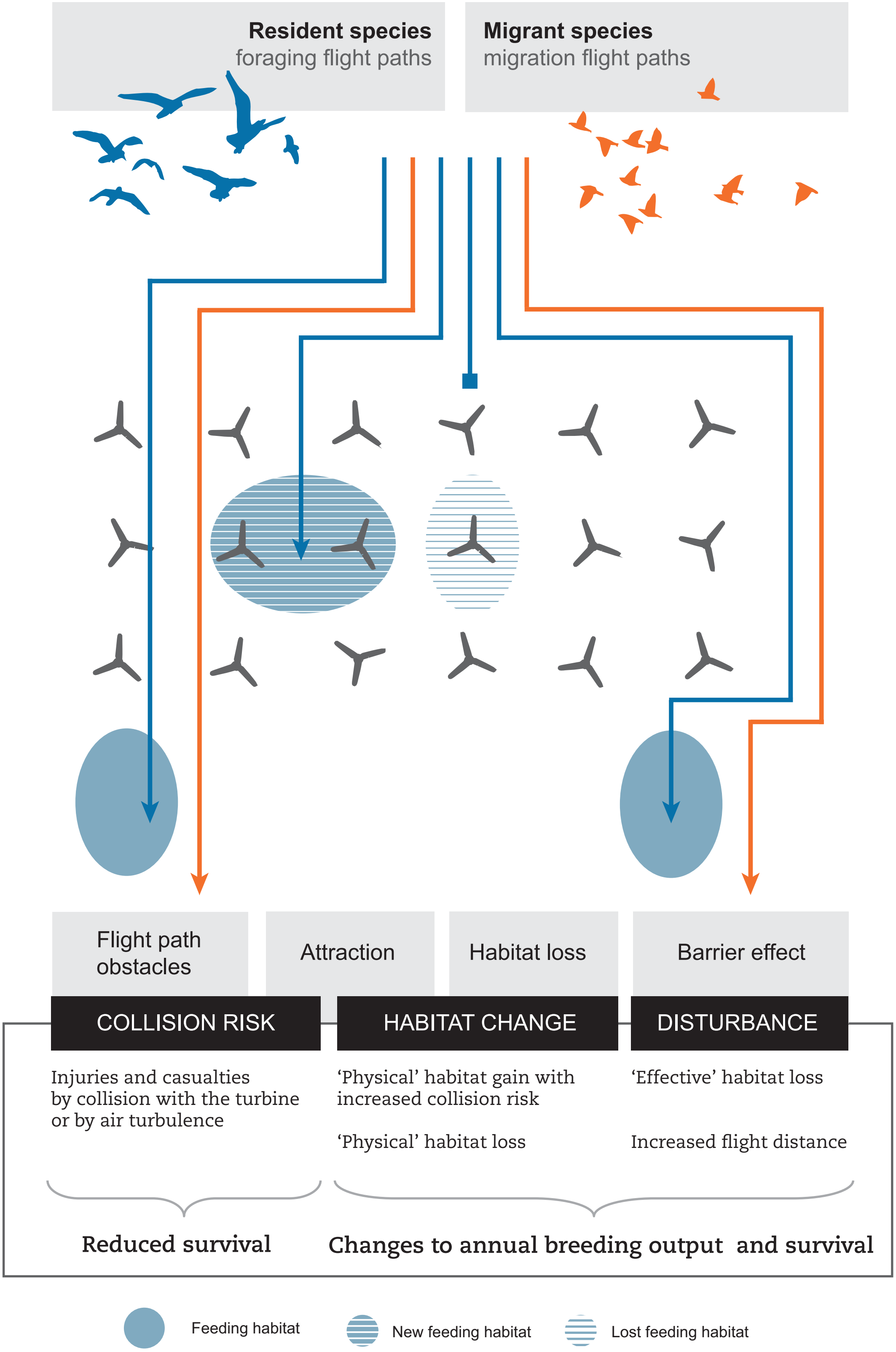


Fig.4 *Conceptual summary describing the three major potential impacts of offshore wind farms on seabirds and their physical and ecological effects.*

However, indirect habitat loss might also occur because of the turbine foundations on the seabed, or changes in habitat use by humans. For instance, construction activity and turbine's distribution may affect the site's hydrology and have an impact over greater areas(Percival, 2003). There is uncertainty about the magnitude of such changes, but the damage may be significant especially on feeding areas such as sandbanks in shallow waters (Drewitt and Langston, 2006).

Turbine bases tend to have a 'reef effect' that increases biodiversity through habitat creation (Linley et al., 2007), but this may influence floral and faunal communities in complex ways generating both positive and negative effects depending on the site and the species (Perrow et al., 2011). Seabirds might also be differently affected by these changes in the habitat. While some specialist species may lose important food sources, other opportunist species (*e.g.* gulls) may increase their presence in the area to exploit the new food source or, as it happens with cormorants, seabirds may simply be attracted to turbine maintenance platforms to use them as perching structures (Kahlert et al., 2004). Nevertheless, this gain of habitat might be counterbalanced by higher collision risk.

Disturbance

The presence of turbines, as well as vessels and people movements related to site construction and maintenance, can potentially deter some seabirds from using areas within and surrounding wind farms. These displacements result in actual habitat loss not because physical changes of the area but as a consequence of a behavioural response. The scale of disturbance effects varies greatly depending on a wide range of factors (Drewitt and Langston, 2006). Site-specific factors depend, for instance, on the offshore wind farm location with respect to important habitats, design of the turbine array and distance between turbines. Moreover seabirds may show different disturbance level depending on their diurnal and nocturnal activity patterns (Desholm and Kahlert, 2005) or different weather conditions. Behavioural responses to the wind farms may vary between species but also between individuals of the same species according to factors such as stage of life cycle (wintering, moulting and breeding), flock size or tendency to habituation.

Even if disturbance and displacement occurs, it may actually be inconsequential if there are abundant alternative habitats. However, offshore wind farms located in migratory flyways or in local flight paths might alter birds' movements increasing their energy expenditure (Masden, Haydon, et al., 2010), this particular type of disturbance is called the 'barrier effect'. In fact, observations in operational wind farms show that many birds chose to fly outside the wind farm rather than fly between the turbines (Desholm and Kahlert, 2005; Larsen and Guillemette, 2007). Unfortunately, there is a lack of complete before-and-after control-impact studies (BACI) in many operational wind farms to properly quantify the barrier and disturbance effects compared to baseline behaviour of seabirds (Drewitt and Langston, 2006).

Gap of knowledge

The internationally agreed guidelines recommend the assessment of collision risk with radar studies in strongly migratory areas (Desholm et al., 2006; Fox et al., 2006; Kunz et al., 2007) and density maps as a proxy to assess the loss of foraging habitats by avoidance or physical habitat modification

(Camphuysen et al., 2004; Fox and Petersen, 2006).

Regarding collision risk assessment, radar technology is a powerful tool to improve our knowledge on spatio-temporal patterns of some seabird groups. Data gathering from radars and the analysis of the outputs require comprehensive studies that already have been addressed in published thesis (Desholm, 2006; Brookes, 2009; Mateos, 2009). In contrast, the use of density maps has fallen behind in the integration of the spatio-temporal dimension of seabird patterns despite seabird distribution maps play a prominent role in most EIA and SEAs assessments. Regarding seabird distribution and abundance, data is usually reported as simple locations or density grids. After a review of more than 200 published studies, Tremblay *et al.* (2009) remarked that "the simple display of distribution data has been much more commonly used than quantitative indices". Indeed, few studies have attempted to address analytical and synthetic methods to extract adequate decisions at strategic (SEA) or local (EIA) levels from seabird distribution data. This thesis aims to contribute to fill in this gap in the methodological approach to the use of seabird distribution data for Offshore Wind Energy Assessments.

OBJECTIVES

“Only when you reach the end of the path,
your footsteps become meaningful”

MAIN OBJECTIVE

The major objective of this thesis was to gain insight into analytical tools in space and time for offshore wind energy environmental assessment in order to provide practitioners with guidelines on how and when to apply them.

SPECIFIC OBJECTIVES

To achieve this objective, this thesis has been structured in four chapters and a global discussion that address the following specific objectives:

1. Design and test a vulnerability index to assess the potential effects of offshore wind energy development on seabirds. (Chapter 1 and 2)
2. Develop a tool to integrate the spatial and temporal variability of seabirds’ abundance at sea to quantify the potential impacts of offshore wind farms on seabirds. (Chapter 3)
3. Demonstrate the limitations of distribution and abundance maps through the individual-based tracking of a flagship species. (Chapter 4)
4. Provide practical guidelines on how to integrate the presented analytical tools in the design of SEAs and EIAs. (Discussion)

SUPERVISORS' REPORT

Dr. Xavier Ferrer, **Dr. David R.Vieites** and **Dr.Grégoire Certain** co-supervisors of the PhD thesis entitled *"Offshore wind energy and birds: integrating assessment tools in space and time"* certify that the dissertation presented here has been carried out by Isadora Christel Jiménez in its totality and grants her the right to defend her thesis in front of a scientific committee.

As supervisors, we have participated in designing, guiding and correcting earlier drafts of the chapters and manuscripts written by the PhD candidate. The contribution of the PhD candidate to each manuscript is detailed below:

CHAPTER 1: A refined methodology to estimate the vulnerability of seabird community to the establishment of offshore wind farms
G. Certain, **I. Christel**, B. Planque and V. Bretagnolle
Journal of Applied Ecology. Impact Factor: 5 (Submitted)
IC: Data analysis and writing.

CHAPTER 2: Wind farm Sensitivity Index for seabirds - Assessing offshore wind energy development on the coasts of the Iberian Peninsula
I. Christel, A. Cama,G. Certain, J.M. Arcos, J. Bécares, B. Rodriguez, SPEA, D.R. Vieites and X. Ferrer
Ecological applications. Impact Factor: 5.1 (waiting for submission)
IC: Analytical study design, data analysis and writing

CHAPTER 3: Seabird aggregative patterns: a new tool for offshore wind energy risk assessment
I. Christel, G. Certain, A. Cama, D. R. Vieites and X. Ferrer
Marine Pollution Bulletin. Impact Factor: 2.5 (Accepted)
IC: Study design, raw observational data processing, data analysis and writing

CHAPTER 4: Foraging movements of Audouin's gull (*Larus audouinii*) in the Ebro Delta, NW Mediterranean: A preliminary satellite-tracking study
I. Christel, J. Navarro, M. del Castillo, A. Cama and X. Ferrer
Estuarine, Coastal and Shelf Science (2012) 96: 257-261. Impact Factor: 2.3
IC: Data collection and processing, analytical study design, data analysis and writing

We also certify that none of the manuscripts included in this PhD thesis has been used as a part of another PhD thesis.

METHODOLOGICAL APPROACH

"Integrative tools: Simplifying ecological complexity"

SEABIRD SURVEYS

From the existing census techniques, the best available methods for obtaining bird distribution and abundance at sea are aircraft and ship-based surveys. Boat surveys have been largely used following a standardised methodology (Tasker et al., 1984) with adaptations according to each particular project. Aerial surveys of seabirds at sea has had a rapid expansion in the last decade and it has been highly influenced by the Danish experience related to the EIA of offshore wind farms (Camphuysen et al., 2004). So far, the methodology explained in their reports (*e.g.* Noer et al., 2000) has become an standard.

The choice between either surveying method depends on the specific research objectives since each method has both advantages and disadvantages (Camphuysen et al., 2004 for a full review). Boat surveys are especially adequate to make exhaustive counts, enabling better species identification with enough time to collect additional information as age, behaviour or flight height. However, this method has two main disadvantages. Firstly, vessels at sea -even if they don't provide food- have an attraction effect on birds which modifies at some degree the original distribution of the seabirds (Spear et al., 2004). Secondly, this method requires longer time at sea to cover large areas. Aerial surveys, on the other hand, are particularly effective in a simultaneous coverage of large areas providing a snapshot of distribution and abundance (Camphuysen et al., 2004) with a minimum attraction or repulsion bias (Certain and Bretagnolle, 2008). Furthermore, aerial surveys can survey distant inaccessible areas (*e.g.* shallow areas or sandbanks) in short time spans with low per-kilometre costs (Camphuysen et al., 2004; Garthe et al., 2011). This is possible thanks to the speed of aircrafts, but this speed is also the main disadvantage of the method. Aerial surveys are performed at the minimum flight speed that ensures flight safety and provides enough observation time (usually 185 km/h). At this speed, there is a short observation time that leads to identification problems for some species, reduced count accuracy and miscounts of rare and small species which are difficult to detect from the aircraft (Camphuysen et al., 2004; Henkel et al., 2007). Moreover, additional information is not always easy to collect and flight height cannot be calculated.

In this thesis both boat and aerial surveys have been used as source datasets of seabird distributions. Seabirds present dynamic and scale dependent distribution patterns hence the datasets to tackle this variability must be easily repeated in similar conditions. Aerial surveys outperform for these spatio-temporal analyses as a particular area can be surveyed several times within a year, and therefore, have been used in the first and third chapter. Boat surveys require more time but maximize the detected species richness (number of individual species or taxa identified on each survey) (Henkel et al., 2007), a key feature to capture detailed biodiversity patterns. In the second chapter the study area covers the coasts of the Iberian Peninsula. Simultaneous and systematically repeated surveys were not economically viable. Therefore, a maximization of species detection through boat surveys was particularly important.

Fig.5 *Picture of one of the vessels used for boat surveys in Chapter 2. (Photo: Pep Arcos)*

Fig.6 *Partenavia P68, airplane model used in the aerial surveys of Chapter 3. (Photo: Albert Cama)*

Both types of surveys aim to monitor a given area to see if seabirds use it, while it seems more intuitive to monitor seabirds to study how they are using an area (Perrow et al., 2006). This move from survey data to tracking data requires a change from population-based studies to individual-based studies and has become possible thanks to the use of electronic-based methodologies, such as satellite tracking transmitters, GPS receivers or radio telemetry. Since the early 1990s, telemetry utilization has constantly increased due to the advances in the miniaturization of the electronic devices (Tremblay et al., 2009).

In the assessment of the offshore wind energy interaction with seabirds, telemetry is an efficacious approach to integrate the spatial and temporal dimension of the distribution patterns of seabirds. Nevertheless there are some drawbacks for this methodology. Some of these devices have high costs; data usually depends on a small sample size; it requires a large amount of analytical time and only a limited number of seabird species can be captured to attach the tagging methods (Perrow et al., 2006). This approach, however, provides fine-scale behavioural studies and could be especially useful if used together with surveying methods such as boat or aerial surveys (Tremblay et al., 2009) and therefore it was used in the fourth chapter of the thesis.

Fig.7 *Satellite tracking device attached to an Audouin's gull (Larus audouinii) (Picture: Isadora Christel)*

STUDY AREAS

This PhD tackles the issue of offshore wind energy development and birds interactions from a methodological perspective with no focus on a particular area. However, in order to present an analytical tool, real data is far better than simulated datasets to understand the tool, its implementation and applicability for decision-making and management. The three study areas belong to French, Portuguese and Spanish waters and have a potential for future offshore wind energy development. Except for the experimental floating turbine in Portugal, so far there is no constructed OWF in any of the study areas, which makes them relevant examples on how to apply the analytical tools for future decision-making. A brief description of the three areas is given below.

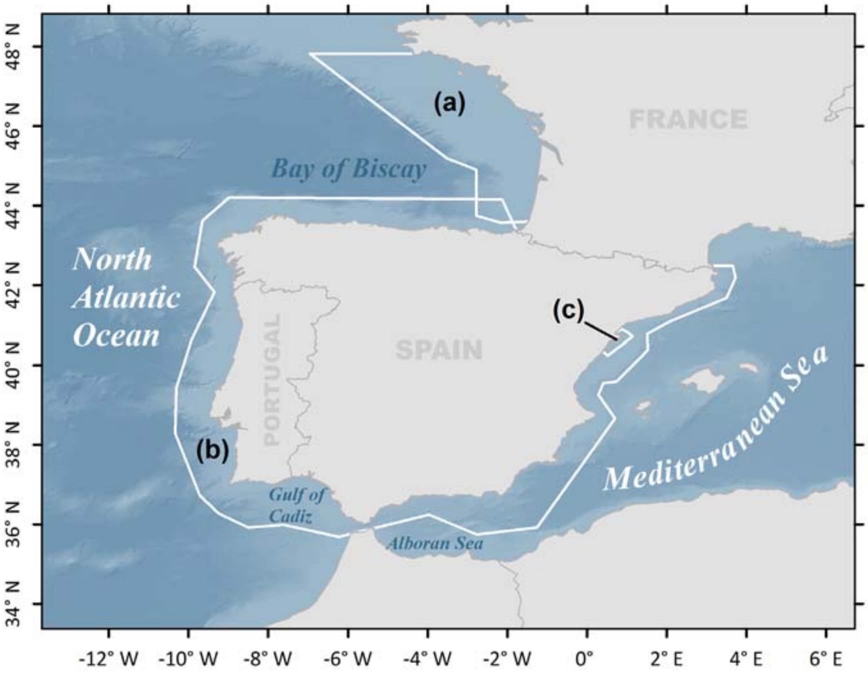
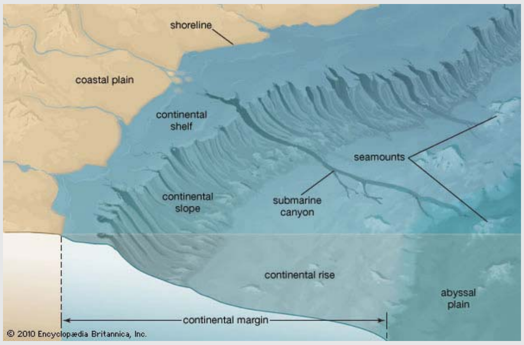


Fig.8 *Study areas:* **(a)** *the French continental shelf of the Bay of Biscay,* **(b)** *the coasts of the Iberian Peninsula and* **(c)** *the Ebro Delta continental shelf*

GLOSSARY



The continental margin: *Submerged prolongation of the continental crust up to the edge of the oceanic crust.*

The continental shelf: *Flat surface with low depth gradient that extends up to the shelf break*

The continental slope / shelf break: *Seaward zone where the seafloor depth gradient increases sharply.*

Box 3 *Glossary and diagram of the Continental margin and its domains. (Definitions by Maestro et al., 2012; Illustration by Encyclopædia Britannica, Inc.)*

Bay of Biscay

The Bay of Biscay is a gulf of the Atlantic Ocean that lies between Cape Ortegal in Galicia, Spain (43.77°N, 7.89°W) and the island of Ushant in Brittany, France (48.43°N, 5.18°W). Within this area, a region of 100000 km^2 (Fig. 8a) was covered with 5000 lineal km of aerial transects on a monthly basis from October 2001 to March 2002.

The study area covered the French continental shelf of the Bay of Biscay between Penmarch in the north (47.75°N, 4.28°W) and Bayonne in the south (43.497°N, 1.64°W). Coastal and shelf break areas are the most productive systems of the region (Certain et al., 2008). The Loire and Gironde river runoffs are a source of nutrient-rich fresh water (Planque et al., 2004) and the shelf break is an area of enhanced primary production as the deep cooler waters reach the euphotic layer due to internal tides and waves (Gerkema et al., 2004), particularly in the southern area that is characterized by a deep canyon, Cap Ferret (Laborde et al., 1999).

The community of seabirds in this area can be classified in six taxonomic groups: Petrels, Gannets, Skuas, Gulls, Terns and Auks (see Table 2 for details on species).

Iberian Peninsula coasts

This area of *ca*. 230000 km^{2}covers the Spanish and Portuguese continental shelf and spans over 7800 km of coastline (Fig. 8b). Boat surveys were carried out by SPEA (the Portuguese Society for the study of birds) and SEO/Birdlife (the Spanish Ornithological Society) in different stages between 1999and2011.

The location of the Iberian Peninsula, surrounded by the Atlantic Ocean and the Mediterranean Sea, and the geomorphological and oceanographical diversity of its continental margins, has significant implications regarding its climate and water mass circulation (Maestro et al., 2012 for a detailed revision). The continental margin of the Iberian Peninsula has well differentiated regions conditioned by many oceanographical aspects like the Eastern North Atlantic Upwelling Region and the Iberian Poleward Current that have a strong influence in the Portuguese, Galician and the Bay of Biscay continental margins(Peliz et al., 2005; Llope et al., 2006); the Mediterranean Outflow Water that flows from the Strait of Gibraltar along the continental slope of the Gulf of Cádiz(Ribas-Ribas et al., 2011); the Modified Atlantic Water that affects the Alboran Sea; and other Mediterranean water masses that influence the Valencia, Catalan and Balearic continental margins(Salat, 1996). This oceanographic settings affect the composition and structure of plankton and all the components of the food web (Santos et al., 2007; Cabal et al., 2008) up to the highest trophic levels and therefore, seabirds. Indeed, the Iberian Peninsula hosts the highest diversity of seabirds in Europe. The community of seabirds in this area has up to 39 usual species from nine different taxonomic groups (Table 3) in addition to rare species that can eventually be found.

Ebro Delta

At a more local scale, the third area was located on the surroundings of the Ebro Delta (40.7° N, 0.75° E; Fig. 8c). The study area covered 1435 km^2 of the continental shelf from l'Ametlla de Mar harbour (24 km North; 40.86° N, 0.8° E) to Peñíscola (51 km South; 40.35° N, 0.4° E). It could be covered in a single day with an aircraft and the aerial surveys were carried out monthly from April 2005 to March 2006.

				BB		IP	ED
Group	**Family**	**Common name**	**Scientific name**	**B**	**A**	**B**	**A**
Petrels	**Procellariidae**	Cory's Shearwater	*Calonectris diomedea*	●	●	●	●
		Northern Fulmar	*Fulmarus glacialis*	●		●	
		Great Shearwater	*Puffinus gravis*	●	●	●	
		Sooty Shearwater	*Puffinus griseus*	●	●	●	
		Balearic Shearwater	*Puffinus mauretanicus*	●	●	●	●
		Manx Shearwater	*Puffinus puffinus*	●	●	●	
		Yelkouan Shearwater	*Puffinus yelkouan*	●		●	
	Hydrobatidae	European Storm-petrel	*Hydrobates pelagicus*	●		●	●
		Wilson's Storm-petrel	*Oceanites oceanicus*			●	
		Madeiran Storm-petrel	*Oceanodroma castro*			●	
		Leach's Storm-petrel	*Oceanodroma leucorhoa*			●	
Cormorants	**Phalacrocoracidae**	European Shag	*Phalacrocorax aristotelis*	●		●	
		Great Cormorant	*Phalacrocorax carbo*	●		●	●
Gannets	**Sulidae**	Northern Gannet	*Morus bassanus*	●	●	●	●
Skuas	**Stercorariidae**	Great Skua	*Catharacta skua*	●	●	●	●
		Long-tailed Jaeger	*Stercorarius longicaudus*	●	●	●	
		Parasitic Jaeger	*Stercorarius parasiticus*	●	●	●	●
		Pomarine Jaeger	*Stercorarius pomarinus*	●		●	
Gulls	**Laridae**	Herring Gull	*Larus argentatus*	●	●	●	
		Audouin's Gull	*Larus audouinii*			●	●
		Great Black-backed Gull	*Larus marinus*	●	●		
		Lesser Black-backed Gull	*Larus fuscus*	●	●	●	●
		Slender-billed Gull	*Chroicocephalus genei*			●	●
		Mediterranean Gull	*Larus melanocephalus*	●	●	●	●
		Yellow-legged Gull	*Larus michahellis*	●		●	●
		Little Gull	*Hydrocoleus minutus*	●		●	
		Black-headed Gull	*Chroicocephalus ridibundus*	●		●	●
		Sabine's Gull	*Xema sabini*	●		●	
		Black-legged Kittiwake	*Rissa tridactyla*	●	●	●	
Terns	**Sternidae**	Black Tern	*Chlidonias niger*	●		●	●
		Little Tern	*Sternula albifrons*			●	●
		Common Tern	*Sterna hirundo*	●	●	●	●
		Gull-billed Tern	*Gelochelidon nilotica*			●	
		Arctic Tern	*Sterna paradisaea*			●	
		Sandwich Tern Sterna	*Sterna sandvicensis*	●	●	●	●
Auks	**Alcidae**	Razorbill	*Alca torda*	●	●	●	●
		Atlantic Puffin	*Fratercula arctica*	●	●	●	●
		Common Guillemot	*Uria aalge*	●	●	●	
Seaducks	**Anatidae**	Common Scoter	*Melanitta nigra*			●	
Waders	**Scolopacidae**	Red Phalarope	*Phalaropus fulicarius*			●	

Table 2 *List of species detected in the three study areas. For each study area (BB= Bay of Biscay; IP= Iberian Peninsula; ED= Ebro Delta) a dot indicates the presence of the species in the boat surveys (B) or the aerial surveys (A). In the Ebro Delta aerial surveys the Razorbill and the Atlantic Puffin* (Alcidae) *could not be differentiated and therefore the species were recorded as a unique group.*

This area has a permanent upwelling thanks to the combination of the influence of the Liguro-Provençal-Catalan front, the sudden broadening of the continental shelf and the source of nutrients from the Ebro river runoff (Palomera, 1992; Arcos, 2001). The high productivity of the area supports an important fishing fleet which is a key feeding source for breeding and wintering seabirds in the Ebro Delta (Arcos, 2001; Arcos et al., 2008). Moreover, the Ebro Delta is a wetland of international importance included in the Ramsar Convention since 1993. With 320 km^2, it is the second most important wetland of the western Mediterranean after the Camargue in France and the second most important from the Iberian Peninsula after Doñana. The rice fields, lagoons, salt pans and beaches of the Ebro Delta provide a variety of habitats for breeding and wintering birds but also a stopover point for large numbers of migratory birds. In global, more than 300species of birds can be found in the area(Bigas, 2012); 18 of which could be detected at sea from the aircraft (Table 4).

MODELLING TOOLS

Whether data on the distribution of seabirds at sea can be a useful tool for conservation and environmental assessment depends on whether the spatial data from seabird surveys represent a general pattern or only a punctual ‘snapshot’ of a highly dynamic system (Fauchald et al., 2002).

Despite its superficial homogeneity, the sea is a heterogeneous environment because of its multiple hydrographical characteristics and the patchy distribution of its biota(González-Solís and Shaffer, 2009). The spatial and temporal distribution of animals is the result from the combination of extrinsic processes, related to the influence of biotic and abiotic environmental factors, and intrinsic processes, related to population dynamics and intra-specific interactions (Bellier et al., 2010). Moreover, the spatial and temporal distribution of seabirds is scale-dependent and patchy over a range of spatial and temporal scales (Hunt and Schneider, 1987; Kotliar and Wiens, 1990; Fauchald et al., 2000), which is explained under the hierarchical patch dynamic theory (Kotliar and Wiens, 1990; Allen and Hoekstra, 1991; Wu and David, 2002).

In a hierarchical patch dynamic system, one would expect large-scale patterns to be more stable and predictable because of a high correlation with environmental variables that define a potential habitat (Hunt and Schneider, 1987; Bellier et al., 2010). At smaller spatial scales one might expect less predictable spatial patterns because smaller patches with high densities of organisms are the result of a particular combination of circumstantial variables that create a temporal preferential habitat within the potential habitat (Bellier et al., 2010).

Translating these theoretical concepts to applied ecology, the optimal assessment tools for seabirds-OWF interactions must take into account this differential effect of spatial and temporal scales. At large-scale assessments, the observed distribution patterns can be considered stable in time and a proxy to potential habitats and thus optimal for the demarcation of key areas of protection (e.g. Important Bird Areas, IBAs) and key areas for offshore wind energy development. At regional or local-scale assessments, the observed clustering of seabirds must be evaluated in its full temporal and spatial variability as a preferential habitat and, consequently used to quantify -in terms of probability- the risk exposure to OWF.

Large scale: Sensitivity Index

The Strategic Environmental Assessment integrates data at really broad scales; therefore we can assume that the temporal scale is not a priority whenever data from different years or periods can be pooled. Seabird distributions might have different patterns depending on the stage of life cycle (wintering, migrating and breeding) but in global their distribution is expected to be spatially and temporally predictable (Fauchald et al., 2002). In other words, at strategic levels the main concern regarding seabirds-OWF assessment is the spatial overlap of seabird distribution with key developing areas of OWF. This is usually dealt with the selection of presence/absence maps of a few flagship species expected to be highly vulnerable to OWFs and general density maps with the global numbers of seabird counts at sea. In this context, it becomes appropriate to apply an index to integrate all these information layers into a summarizing one.

Garthe and Hüppop (2004) proposed the Wind farm Sensitivity Index (WSI) to map the vulnerability of seabirds to offshore wind farms in a sea region. This index estimates first the vulnerability of each species according to their sensitivity to collision risk, disturbance and their demographical and conservation status. This value is later combined with the spatial abundance of each species to obtain a vulnerability map.

This method is general, simple and widely applicable, hence instead of developing a new index this thesis examines the method in depth and makes recommendations on the optimal application of the index for its utilization in any Strategic Environmental Assessment.

Regional and local scale

At smaller scales, Environmental Impact Assessments usually focus in seabird habitat-use strategies and the processes that are expected to influence seabird occurrence or the availability of their prey. As it is done at large scales, bird densities are used as a proxy of bird habitat to assess risk exposure to habitat loss or disturbance. Despite this is a common practice, the effectiveness of this method is compromised by the assumption that at these scales the observed data follows a normal distribution. In fact, animal count data is seldom normal. Seabird aerial and boat surveys data are zero-inflated (Broek, 1995; Pearce and Ferrier, 2001; Barry and Welsh, 2002) with a positive skew of non-zero values, *i.e.* many counts of low to intermediate density and very few counts of high density (Fauchald et al., 2002; Mcsorley et al., 2005; Certain et al., 2007). For this reason, the explicit consideration of temporal and spatial variability of seabird occurrence and density is necessary in any EIA to design ecologically sound management strategies at regional and local scales (Tobin, 2004; Certain et al., 2007).

The third and fourth chapter of this thesis tackle this spatio-temporal variability through the application of Taylor's Power Law and the analysis of individual's movements respectively.

Aggregative response

The first method is based on an empirical relationship that expresses the dependency between the average measured in one point and the variance of the measures in this point(Taylor, 1961).Although its mathematical foundations have been widely discussed (Kendal, 2004), this relationship has been demonstrated for more than 400 species in taxa ranging from protists to vertebrates (Kilpatrick and Ives, 2003) and it is true for both spatial data (repeated measures adjacent in space) and temporal data (repeated measures in a point over time) (Taylor and Woiwod, 1980, 1982; Taylor et al., 1980). When calculated through space, Taylor's Power Law provides a measure of the strength of the aggregative response of organisms (Jiménez et al., 2001; Östman, 2002). When calculated through time, it can be used as an index of the temporal variability of the spatial distribution of organisms, highlighting recurrent and occasional presence areas (Certain et al., 2007). Therefore this method provides a useful framework to study the spatio-temporal variability in seabird surveys.

Individual tracking

The second method deals with spatio-temporal variability from an individual-based perspective. Seabird surveys are constrained in space by the arrangement of the survey transects, and constrained in time by the moment of the day at which each transect is surveyed and the necessity of daylight for the counts. Individual tracking of seabirds, instead, is not restricted in either space or time. When applying multivariate modelling, temporal patterns in space use can be described not only monthly or seasonally but also within a circadian cycle. Moreover, if a transmitter provides frequent locations, it is possible to quantify the geometric properties of the path of a tagged animal (e.g. speed, heading, turning angles) (Patterson et al., 2008). A particular combination of values for these properties can be interpreted as a behavioural mode (feeding, travelling between foraging patches, resting, etc.). Under this assumption, State-Space Models (SSM; Jonsen et al., 2003) can be applied to calculate the probability of an animal being in a particular behavioural mode and later individual decisions can be linked to population distribution and applied to risk assessments (Turchin, 1998).

Research papers

RESUM

L'energia eòlica marina és una de les fonts d'energia renovable més prometedores per al futur. No obstant això, l'establiment d'aquestes instal•lacions requereix una avaluació d'impacte detallada, en particular pel que fa a les poblacions d'aus marines. Fins on sabem, l'únic marc de treball disponible a gran escala que permet aquesta avaluació és el plantejament desenvolupat al 2004 per Garthe i Hüppop. Conceptualment es tracta d'un treball molt sòlid, però el tractament matemàtic dels conceptes no es correcte i cal que sigui actualitzat. L'estudi que es presenta en aquest capítol revisa el treball de Garthe i Hüppop destacant els supòsits en els que es fonamenta i els problemes d'interpretació associats als mateixos. Fet això, fem una reestructuració exhaustiva del marc matemàtic fent que sigui correcte tant en el seu aspecte formal (matemàtic) com en la seva interpretació ecològica. D'aquesta forma l'índex, ja de per sí molt últil es torna més adaptable i pràctic. La revisió que es presenta en aquest treball diferencia explícitament el risc de col•lisió i el risc de pertorbació; es basa en els desenvolupaments teòrics més recents d'ecologia de comunitats; i proposa una integració seqüencial dels efectes des d'un nivell d'espècie fins al de comunitat. Mitjançant el cas d'estudi de les aus marines del Golf de Biscaia (França) il•lustrem les limitacions del plantejament anterior i la utilitat de la nostra revisió de l'índex anterior. En general, el marc refinat proporciona informació clara, complementària i sense ambigüitats que ha d'ajudar als gestors de l'àmbit marí en la pressa de decisió sobre les localitzacions òptimes per als parcs eòlics marins i l'avaluació dels possibles impactes que es pot espera en determinades zones. A més, el mètode a través del qual integrem la vulnerabilitat de les espècies a nivell de la comunitat és de caire general, i podria ser fàcilment adaptat a qualsevol tipus d'impacte i comunitats animals més enllà del cas particular de les aus i l'energia eòlica marina

JOURNAL REFERENCE

Paper submitted to the *Journal of Applied Ecology*

A refined methodology to estimate the vulnerability of seabird community to the establishment of offshore wind farms

1

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Portugal.

ABSTRACT

Marine offshore wind farms are amongst the most promising renewable energy sources for the future. However, their proper establishment requires thorough impact assessment, in particular with regard to seabird populations. To our knowledge, the only available framework for such assessment is the approach developed in 2004 by Garthe&Huppöp. Although conceptually sound, the approach is mathematically incorrect and needs to be up-dated. This study briefly reviewsGarthe & Hüppop's approach, highlighting the hidden assumptions and interpretation problems associated to it. Then, we deeply refined the framework by making it ecologically and mathematically sound, tractable, and adaptable. The refined approach explicitly disentangles collision and disturbance risk,draws upon recent theoretical development in community ecology and proposes a sequential integration of the impact at the species and community level. We illustrate the pitfalls of the previous approach and the usefulness of our refined framework through a case study on the seabird populations over the continental shelf of the Bay of Biscay, France. Overall, the refined framework provides clear, complementary and unambiguous information to managers about the localization and the kind of impact to be expected. Furthermore, the method through which we integrate vulnerability from species to community level is general, and could easily be transposed to any kind of impact and communities.

INTRODUCTION

The development of marine offshore wind farms has increased significantly worldwide in the last decades, following the need of decreasing carbon footprint through the exploitation renewable energy source (Punt et al., 2009). Ecological effects of locating wind farms offshore can be both detrimental and beneficial (Punt et al., 2009). Among the detrimental effects, wind farms are potential treats to marine seabirds in two different aspects: increased mortalities due to collision risk, an increased energy expenditure and habitat loss through disturbance (Exo et al., 2003; Garthe and Hüppop, 2004; Drewitt and Langston, 2006; Masden et al., 2010).

Spatial planning of wind farms requires quantitative assessment of these threats. Garthe & Hüppop (2004) proposed a methodological framework to map the vulnerability of seabird community to wind farm, combining two sources of information. The first is an estimate of the vulnerability of each population in the study area, based on behavioural and demographical traits and on conservation status. The second is the spatial distribution of each population, based on extensive at-sea surveys. Although the method is general, simple, and widely applicable, the mathematical formulation contains hidden assumptions that might be problematic and might lead to incorrect estimates of vulnerability as well as biased identification of key areas. In particular, collision and disturbance risk are related multiplicatively and mixed together, different risks factors are given equal weight even though some are related to the risk itself while others are only aggravation factors, and the vulnerability index is weighted by population abundance.

In this study, we briefly review the original approach (Garthe and Hüppop, 2004) and point towards its weaknesses. Then, we attempt to solve the problems by proposing a new methodological approach that (1) explicitly distinguish between risk factor and aggravation factor, (2) allows the separation between collision risk and disturbance risk, and (3) draws on recent development in functional diversity (Leinster and Cobbold, 2012) to produce and map a community vulnerability index based on the local relative frequencies of species within the seabird community. Finally, we apply the refined methodology to the Bay of Biscay, using data collected during an extensive seabird aerial survey in the Bay of Biscay, France (Bretagnolle et al., 2004; Certain et al., 2007; Certain and Bretagnolle, 2008).

METHODS

Reviewing the Wind farm Sensitivity Index (WSI)

The WSI proposed by Garthe and Hüppop (2004) has been successfully implemented to detect areas where the seabird community would be most vulnerable to the establishment of a wind farms (Garthe and Hüppop, 2004). It is based on a Seabird Sensitivity Index (SSI), thought to reflect the vulnerability of each seabird species to the establishment of offshore wind farms, and the at-sea abundances of each seabird species (A). Let us consider an area discretized in a succession of j = 1…L locations and populated by a set of i = 1…S seabird populations. We can write:

$$WSI_j = \sum_{i=1}^{S} \mathrm{h}\left(A_j + 1\right) \times SSI_i \qquad (1)$$

$$SSI_i = \frac{f_{1i} + f_{2i} + f_{3i} + f_{4i}}{4} \times \frac{f_{5i} + f_{6i}}{2} \times \frac{f_{7i} + f_{8i} + f_{9i}}{3}$$

(2)

Name	Impact type	Type of risk factor	Short description
f1	collision	Primary	% time spent flying
f2	collision	Primary	% time spent at high altitude when flying
f3	collision	Aggravation	Flight manoeuvrability
f4	collision	Aggravation	Nocturnal flight activity
f5	disturbance	Primary	Disturbance by ship and helicopter traffic
f6	disturbance	Aggravation	Habitat flexibility
f7	sensitivity	Primary	Biogeographical population size
f8	sensitivity	Aggravation	Adult survival rate
f9	sensitivity	Primary	European Conservation status

Table 1 *Risk factors according to which species vulnerability to windfarm is assessed. Detailed definition for each risk factor can be found in Garthe & Huppop 2004*

Where the set of f_{ir} represent r = 1…9 risk factors for the ith seabird species. These risk factors can be grouped into three risk types: risk related to collision (r = 1, 2, 3, 4), risk related to disturbance (r = 5, 6) and risk related to the overall sensitivity of species (r = 7, 8, 9). Table 1 synthesizes the different risk factors, each being measured on a relative scale ranging from 1 (minimum risk) to 5 (maximum risk). As it is formulated, the current estimate of SSI and WSI makes the following assumptions:

A1: All the risk factors associated to a given risk type are equally weighted, and the relationship between the risk factors of a given risk type is additive.

A2: Each risk type is equally weighted, and the relationship between risk types is multiplicative.

A3: The local importance of a given seabird species in the local measure of the vulnerability of the seabird community is proportional to its local log abundance.

These assumptions might be difficult to hold in a number of cases. First, for a given risk type, risk factors might not be independent, nor additive. We can distinguish two categories of risk factors: those that are directly associated to the risk itself (i.e. time spent flying and time spent at high altitude in the case of the collision risk) and those that are aggravation factors of the risk (i.e. flight manoeuvrability and nocturnal activity). That conceptual difference is important: aggravation factors are not important in themselves, but they can increase a risk that already exists. Following that thread of reasoning, disturbance by ship and helicopter traffic (f5) can be viewed as the real risk factor, while habitat flexibility (f6) only matters if the species is disturbed in the first instance. Finally, biogeographical population size (f7) and European conservation status (f9) both determine the overall sensitivity of a species to any kind of impact, while adult survival rate (f8) correlates to its capacity to replenish the population if some increased mortality is experienced. Therefore, f8 can be viewed as an aggravation factor. If we recognize that risk factors are not of the same kind, but that some hierarchy can be found, in terms of primary risk factors and aggravation factors, then the mathematical formulation of SSI should be adapted to take into account the potential caveats from assumption A1.

Assumption A2 is complex, as it sets on an equal foot collision risk, disturbance risk and overall species sensitivity, and it furthermore assumes that they interact multiplicatively. Measuring the relative importance of collision risk over disturbance risk might indeed be difficult and considering them as equal by default is perfectly understandable. Having a multiplicative relationship between species overall vulnerability and both collision and disturbance risk is also perfectly understandable: the multiplication ensures that the risk is “weighted” by the overall sensitivity of each species. However, having a multiplicative relationship between collision and disturbance is much less understandable, as they do not really depend on each other but are in fact two aspects of the impact that act independently, have different consequences and might lead to very different management measures. If the collision risk is high but the disturbance risk is low, the resulting risk will be much more lowered with a multiplicative relationship than with a simple additive relationship. Furthermore, both collision risk and disturbance risk are completely different in nature, which poses the question of the usefulness of their combination. Informed decision for management could as well be taken simply upon the examination of both collision risk maps and disturbance risk maps. The multiplicative relationship between collision and disturbance risk is therefore questionable, and we propose either to consider these two risks independently or alternatively to use an additive relationships, should they be combined.

Assumption A3 will give more weight to species locally abundant. The intuitive idea behind this is simply to prevent the installation of wind farms in areas where seabirds aggregate. However, applying the weight at the species level, and using the log-abundance of seabirds instead of their abundance can be criticized. Applying the weight at the species level introduces a confusion within the framework, because rare species are first up-weighted in the SSI through the factors f7 and f9, and then down-weighted in the WSI because rare species present probably lower local abundance. Conversely, the importance of abundant species will be first down-weighted in the SSI, and then up-weighted in the WSI. With the current formulation, there is no control on the magnitude of up-weight / down-weight that each species will experience, which renders any interpretation of the spatio-temporal variations of the WSI very difficult. Finally, the use of log abundance simply assumes that the importance of a single seabird in a location decreases exponentially as the total number of seabird in that location increase. A single individual in the middle of nowhere will have more weight, in proportion, than an individual located in a flock of one hundred seabirds. This assumption has neither ecological nor management support.

To take into account the potential caveats associated to assumptions A1, A2 and A3, we propose in the following section a refinement of the WSI framework.

Definitions

In the following development, measures of risk and vulnerability will successively cross three levels of organisation: individuals, species, and community. It is therefore useful to provide some clear definitions and point toward the level of organisation at which they apply. In our context, the word species is fairly equivalent to the word population because most impact assessment are concerned with a delimited area and focus on the population of species within that area, not on the whole bio-geographical distribution of the species.

We will refer to a risk as a measure of the probability that an individual of a given species suffers a given impact. For example, the collision risk refers to the probability that an individual of a given species collides with a wind-farm. We will use the term sensitivity to refer to the overall sensitivity of a given species to any kind of impact. We will use vulnerability when the individual risk of suffering a given impact is integrated at a higher organisation level. We will distinguish two levels: species vulnerability and community vulnerability.

Individual risk and population sensitivity will be estimated as a function of factors, i.e. quantitative or semi-quantitative elements supposed to measure one of the aspects of the risk or sensitivity considered. We will distinguish between primary factors, i.e. factors directly controlling the risk or the sensitivity, and aggravation factors, i.e. factors that contribute to increase an already existing risk or sensitivity. All factors, risks, sensitivity and vulnerabilities measures will be expressed as relative probabilities, ranging between 0 and 1. A value of 0 is interpreted as no risk or no sensitivity or no vulnerability, and a value of 1 is interpreted as maximum risk or maximum sensitivity or maximum vulnerability.

As factors, risks, sensitivity and vulnerability are all expressed on the same scale, they can be conveniently combined through either averaging or multiplication. We will use averaging when the values to be combined do not interact. We will use multiplication when the values to be combined interact.

Combining factors with a power function

Let us denote r the relative estimate of a given risk, and let us assume that r is the combination of two factor: a primary risk factor, a, and an aggravation factor, g. We propose to link r to a and g through the following relationship:

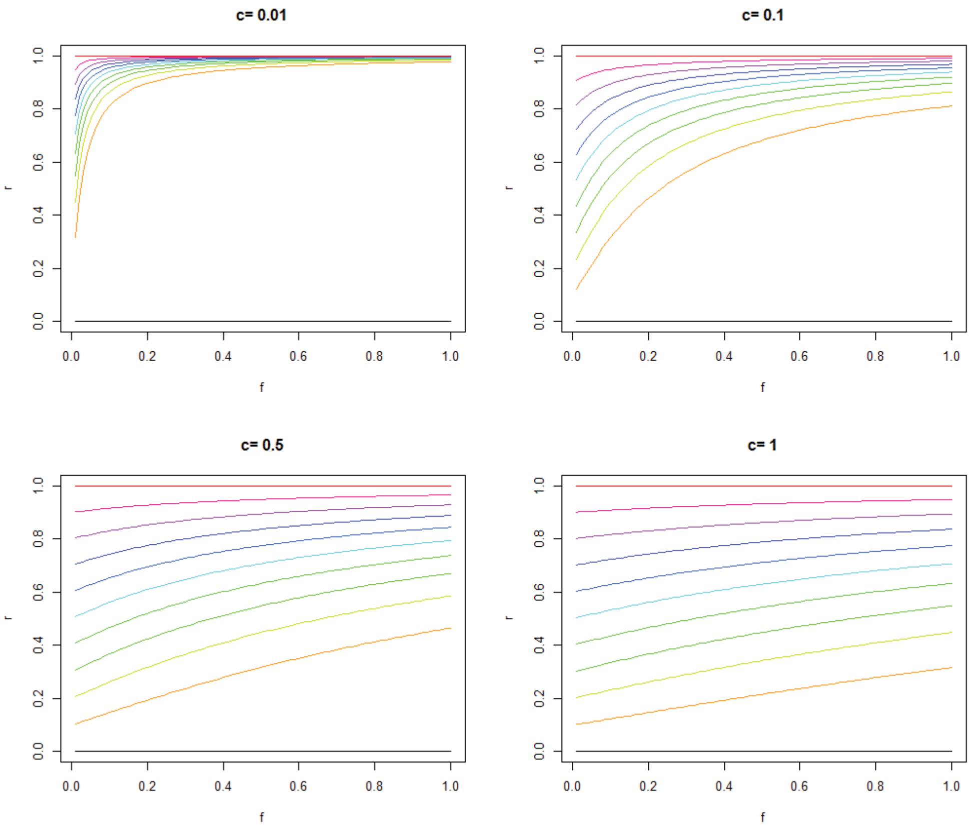

Fig.1 *This figures shows how r (y-axis) changes according to g, for various values of c. In this figure, a = r when f = 0. It is clear that if c is low, then r is strongly dependent on f, whatever a (the starting point of the curve). On the other hand, g has much lower effect on r if c is high.*

$$r = a^{1-\left(\frac{g}{g+b}\right)} \quad , \begin{cases} 0 \le a \le 1 \\ 0 \le g \le 1 \\ 0 \le c \le \infty \end{cases} \qquad (3)$$

Under this formulation, a=r when g=0, and then progressively increases as g increases. The parameter b can be viewed as a measure of the influence of g over r: the smaller the c, the more influence g will have on r (fig 1). We suggest to use c=0.5 as a default and strongly recommend avoiding too small c values (i.e. <0.1, fig 1). Under this formulation, r remains bounded between 0 and 1.

Applying the power risk function

Let us denote the collision risk for the ith species ci, the disturbance risk di, and the species sensitivity si. To obtain an estimate of each, we rely on the set of estimated risk factors for each species ff=(1...9)i, . These are basically produced as values ranging between 1 and 5 (Garthe & Huppop 2004), but as a and g (eq. 3) should be comprised between 0 and 1, they need to be rescaled (i.e. divided by 5).

The collision risk can be seen as the combination of the 4 first factors f1i to f4i. % time spent flying (f1i) and % of time spent at the wind farm altitude when flying (f2i) can be seen as primary risk factors. Manoeuvrability of species (f3i) in flight and nocturnal activity (f4i), on the other hand, can be seen as aggravation factor. ci is therefore obtained by applying eq. 3 with $a_i = f_{1i} f_{2i}$ and $g_i = \left(f_{3i} + f_{4i}\right)/2$. We use a multiplicative relationship between f1i and f2i because we assume they interact. We use an additive relationship between f3i and f4i because we assume they do not interact.

The disturbance risk di can be seen as the combination of a primary risk factor, the intensity of the behavioural response to anthropic activity (f5i), and an aggravation factor, the flexibility of habitat use (f6i). Therefore, di is obtained by applying eq. 3 with ai=f5i and gi=f6i.

The species sensitivity si can be seen as the combination of the 3 last factors f7i to f9i. The biogeographical population size (f7i) and the species conservation status (f9i) can be seen as non-interacting primary risk factors. The natural survival rate of the species (f8i) can be seen as an aggravation factor. Therefore, si is obtained by applying eq. 3 with ai=f7i+f9i and gi=f8i

Moving from risk to vulnerability

Once c_i, d_i and s_i have been defined, they can be combined to get an estimate of the overall species vulnerability v_i to windfarm. Recall that c_i and d_i are individual risks, while s_i is the species sensitivity. We propose to view the vulnerability of a species to a risk as the product of the individual risk by the species sensitivity. Then, we propose to view the overall species vulnerability as a weighted mean of all its risk-specific vulnerabilities. In the context of seabird-wind farm, this lead to the following expression:

$$v_i = \alpha_c \times c_i \times s_i + \alpha_d \times d_i \times s_i$$

$$\text{, with } \alpha_c + \alpha_d = 1 \qquad (4)$$

αc and αd are risk-specific weights controlling the influence of each risk. In our case, setting αc = αd = 0.5 means that vulnerabilities to collision and disturbance are equally weighted. In our framework, vi is the direct equivalent of the SSIi (eq. 2).

The next step is now to integrate the vulnerability of several species into a measure of the vulnerability of a whole community, as it was originally attempted through the WSIj (eq. 1). There, we build upon the recent development of Leinster and Cobbold (2012) that modified the classical estimate of Hill's diversity (Hill, 1973) to take into account species similarity:

$$^{q}Div_{z}(p)=\left(\sum_{i=1}^{S}p_{i}\left(Z_{p}\right)_{i}^{q-1}\right)^{\left(\frac{1}{1-q}\right)}$$

, with $0 \leq q \leq \infty$ (5)

Where pi is the relative frequencies of the ith species, and $\left(Z_p\right)_i$ is a measure of the similarity between an individual of the ith species and an individual taken at random in the community. $\left(Z_p\right)_i$ is expressed between 0 (completely dissimilar) and 1 (identical) and is usually measured through a set of traits for each species, as in classical functional diversity studies (Leinster & Cobbold 2012). This index produces a diversity measures in effective species number, that is the number of equally abundant species required to obtain the same diversity measure. This is recommended practice as it greatly eases the interpretation of the index (Tuomisto, 2010; Leinster and Cobbold, 2012). The introduction of the term $\left(Z_p\right)_i$ gives more weight to the highly dissimilar species. Therefore, simply replacing $\left(Z_p\right)_i$ by 1-vi will produces a diversity measures that gives more weight to the most vulnerable species. Both the formulations of Hill (1973) and of Leinster & Cobbold (2012) contain a parameter, q, which controls the sensitivity of the diversity metric to the weighting parameter, i.e. $\left(Z_p\right)_i$ in the case of Leinster & Cobbold (2012). The greater is q, the higher is the weight of similar species over dissimilar ones. In the vulnerability context we wish to introduce, the quantity 1-vi will be close to zero when the species are highly vulnerable, and we precisely wish to give maximum weight to the most vulnerable species. We therefore set q=0. In a spatial context where community data are available over j = 1…L locations, the overall community vulnerability to wind-farm is written:

$$V_j = \sum_{i=1}^{S}\frac{p_j}{1-v_i} \qquad (6)$$

V_j provides an estimate of the vulnerability of a community to a given impact, in effective species number. It can be interpreted as the number of equally abundant and fully vulnerable species that composes the community. This formula can also be used to measure the community vulnerability to a given risk. Substituting c_i to v_i lead to a measure of the vulnerability of the community to collision, and substituting d_i to v_i lead to a measure of the vulnerability of the community to disturbance. That way, the overall community vulnerability map can be partitioned into each risk component.

Taking Abundances into account

In the original framework, the SSIi was multiplied by log(Aij) where Aij stands for the abundance of each seabird species at each locations, and the sum over the species was taken (eq.1). This unfortunately led to interpretation confusion, especially because the information concerning abundance and species composition are mixed together. On the contrary, Vj fully account for species composition and leaves abundance aside. The total seabird abundance at each location A.j is therefore a natural complement to Vj. It can be computed from survey data and should be systematically presented together with Vj.

Case study: Seabirds populations in the Bay of Biscay.

We applied both the original and refined framework to seabird populations on the continental shelf of the Bay of Biscay, France, that have been extensively sampled though a series of aerial ('ROMER') and ship-based ('PELGAS') surveys (Bretagnolle et al., 2004; Certain and Bretagnolle, 2008; Certain et al., 2011). These surveys resulted in a succession of studies focusing on spatial structure, variability, and prey-predator relationships (Certain et al., 2007, 2011; Bellier et al., 2010, 2012; Chadœuf et al., 2011). Details on both survey methodologies can be found in Certain (2007), Certain & Bretagnolle (2008) and Certain et al. (2011). Briefly, during ROMER, strip-transect aerial surveys covered repeatedly the Bay of Biscay in winter, from October 2001-to March 2002, offering a first exhaustive snapshot of the extent and abundance of the wintering population of seabirds in the Bay of Biscay. Then, from 2003 onward, observers recorded top predator data on board of the RV-THALASSA during the PELGAS cruises that occur each spring in the Bay of Biscay. In both cases, sampling scheme is systematic, constituted of perpendicular transects lines separated by ~20km of each other. Seabird observations are collected continuously along the transect, including species identification and number of individuals. The sampling design covers homogeneously the entire study area (100 000 km2). For data processing, the transects are sliced into 20km of equal sized segments, within which the relative abundance of each species, i.e. number of counted individuals, is reported. To ease the comparison with the previous work, we use the whole ROMER dataset and the PELGAS dataset from 2003 to 2008.

Based on ROMER and PELGAS records, we first established the list of the 30 seabird species encountered and identified in the Bay of Biscay (table S1). We also defined 7 groups for the unidentified observations, together with assumed proportions based on identified sightings (table S1). For each species, the risk factors ff=(1…9)i identified by Garthe & Hüppop (2004) were documented (table S1). If possible, we used the values already documented by Garthe & Hüppop (2004), otherwise we scored the species according to discussion carried out during expert meetings. For groups of unidentified seabirds, we used average values, weighted by species proportions in each group (table S1). Based on the risk factors, ci, di and si were computed for the 37 species and the 7 groups. Then, overall species vulnerability vi was computed, as well as the original SSIi. To reveal how vi differs from SSIi, we looked at the difference between the species rank according to SSIi and the species rank according to vi. We computed correlations between differences in rank and ci, di and si to search which risk was responsible for the observed differences.

Producing diagnostic panels for the Bay of Biscay

The refined framework we develop is not supposed to produce one single map, but rather to produce a few interpretable maps, each capturing one key element to be considered to assess the impact of offshore wind farm on seabird populations. We propose to use diagnostic panels composed of 4 maps, each related to a specific component of the potential impact. The two first maps would present the two distinct and fundamental elements of the impact assessment: the overall vulnerability of seabird community, Vj, and the total seabird abundance map A.j. Then, to further inform management, we show the two risk-specific component of Vj, namely the vulnerability to collision and the vulnerability to disturbance. These two last maps highlight how both risks contribute to the overall community vulnerability. To present the four maps, geostatistical interpolation (Cressie, 1993; Pebesma and Wesseling, 1998) and kriging were carried out to ease the representation and interpretation of the spatial patterns. We interpret these maps in the context of wind-farm impact assessment, and propose locations on the continental shelf were the impact on seabird populations would be minimised.

Comparing the diagnostic maps with the previous WSI

To illustrate the differences between the original and refined approach, we also computed the original WSIj maps, together with simple summed log abundance maps (i.e. removing SSIi in eq. 1). Showing both illustrates how taking into account differing vulnerability for each species modify the perception of the potential impact of the establishment of an offshore wind farm in the original framework.

RESULTS

Comparison between SSI_i and v_i

For the sake of comparison, both SSIi and vi were scaled between 0 and 1. When plotting these scaled values against each other, it is clear that the two indexes generally agrees on the classification of species (fig 2a), even though nearly all species are located above the 1:1 line, suggesting that on average, a seabird species is considered more vulnerable by the vi than by the SSIi. This is better understood in the light of fig 2b, where we see that the scaled distribution of the two indices differs, with the distribu-

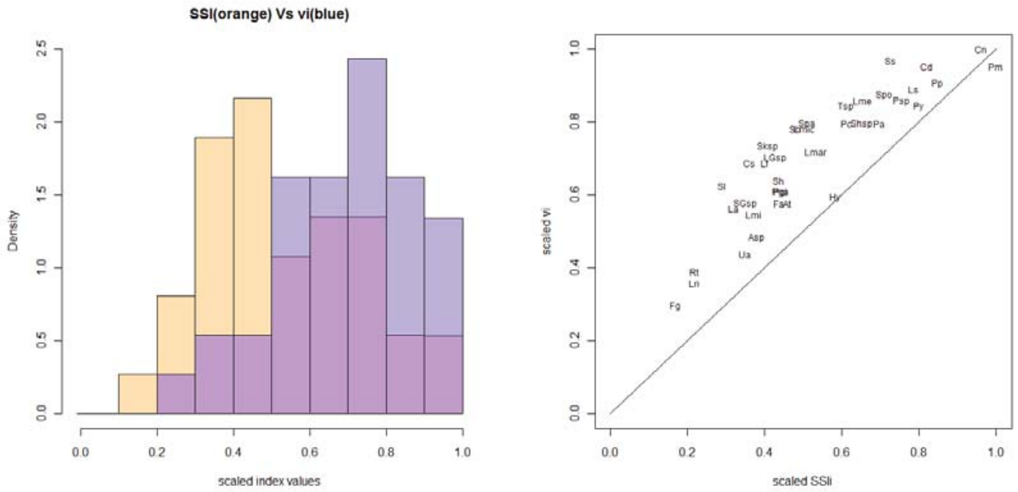

Fig.2 *Comparison between the original SSI and the proposed vi*

tion of scaled vi being closer to 1.

Some species were more affected than others by the change of indices (table S1, last column). For example, skuas are considered more vulnerable with vi than with SSIi, with a difference in rank between -2 and -10, depending on the species. On the contrary, auks are considered less vulnerable with vi, with a difference in rank between 2 and 9 (table S1). The correlation between the differences in the ranking of species according to both metric (table S1, last column) and the component of vi was positive with the collision risk ci (0.45, p=0.005, df=35), negative with the disturbance risk (-0.42, p=0.009), and non significant with species sensitivity (p=0.38).

Diagnostic Maps for Seabirds in the Bay of Biscay

Fig 3 and 4 present diagnostic panels for seabirds in the Bay of Biscay based on ROMER and PELGAS data, respectively. The ROMER-based panel highlight two main area where the seabird community is the most vulnerable, i.e. the northwest area and the south-eastern area (fig 3a). In addition, the wintering population of birds is widely spread in the Bay of Biscay, leaving only few areas where seabird abundance is low (fig 3b). Furthermore, the ROMER panel clearly shows that vulnerability to collision and disturbance differs in space, highlighting that collision-vulnerable communities are distributed further from the coast than dis-

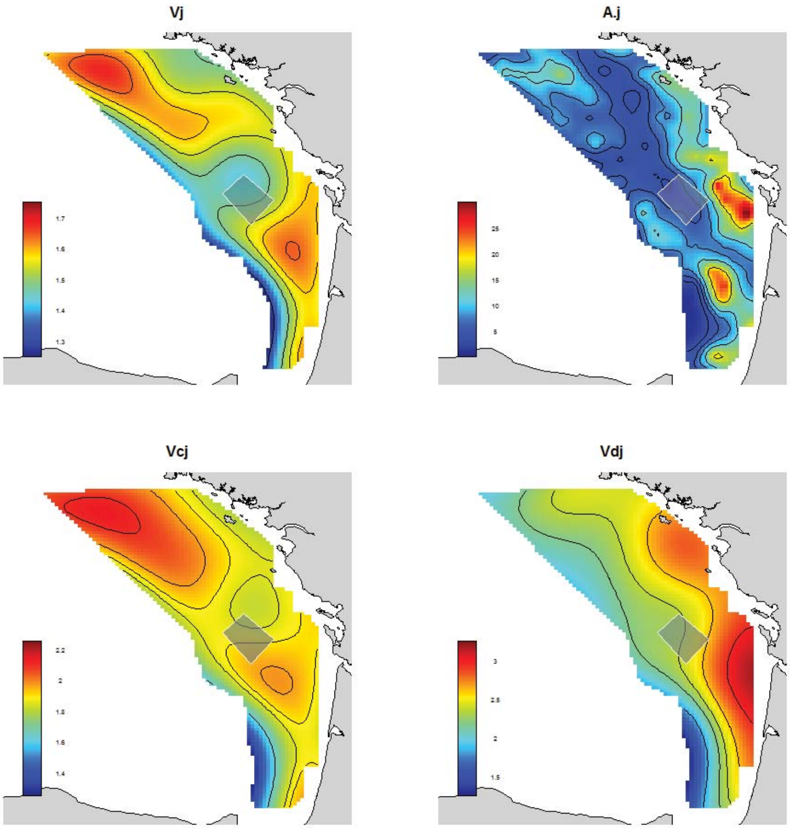

Fig.3 *ROMER-based diagnostic panel. Up left: overall vulnerability of the seabird community. Up-right: toital abundances. Low left: vulnerability to collision of the seabird community. Low right: vulnerability to disturbance of the seabird community.*

turbance-vulnerable communities.

The inspection of PELGAS-based panel offers a slightly different picture. The location of high and low vulnerability areas are roughly the same (fig 3a and 4a), apart from a localised patch of high vulnerability in the North East, around Belle-Ile en Mer, that was not visible from the ROMER-based panel. The abundance map differs more clearly (fig 4b), with high abundances more restricted to the Northernmost and coastal areas. Finally, vulnerability to collision (fig 4c) and to disturbance (fig 4d) presented a rather similar pattern, even though vulnerability to collision is much more spread than vulnerability to disturbance.

The examination of ROMER and PELGAS diagnostic maps reveal some differences between the wintering and spring situations, however, in each case, the central part of the Bay of Biscay, identified in fig 3 and 4 as a grey rectangle, is characterised by low abundances, and low-to moderate vulnerability. As a synthetic result of this impact assessment, we suggest this area as an informed choice for the location of offshore wind farm, as it seems to minimize the impact in both ROMER and PELGAS situations.

Comparing Diagnostic panels with WSI_j maps

Fig 5 displays WSI maps as proposed by the original framework. They globally provide consistent information with the refined framework, but with more emphasis on the disturbance risk than on the collision risk. Furthermore, the spatial patterns displayed by the WSIj (Fig 5a&b) are extremely similar

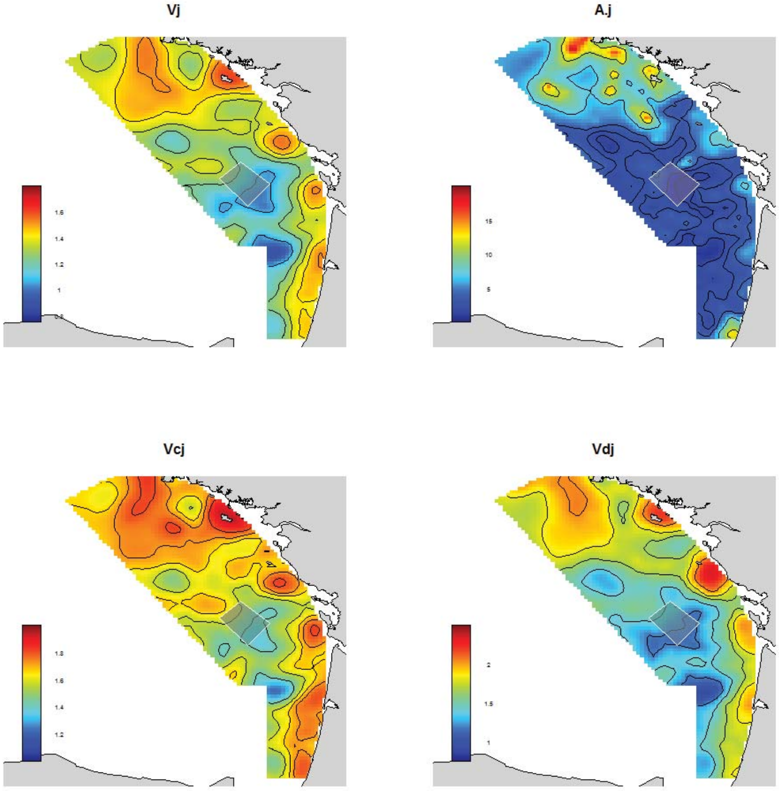

Fig.4 *PELGAS-based diagnostic panel. Up left: overall vulnerability of the seabird community. Up-right: toital abundances. Low left: vulnerability to collision of the seabird community. Low right: vulnerability to disturbance of the seabird community.*

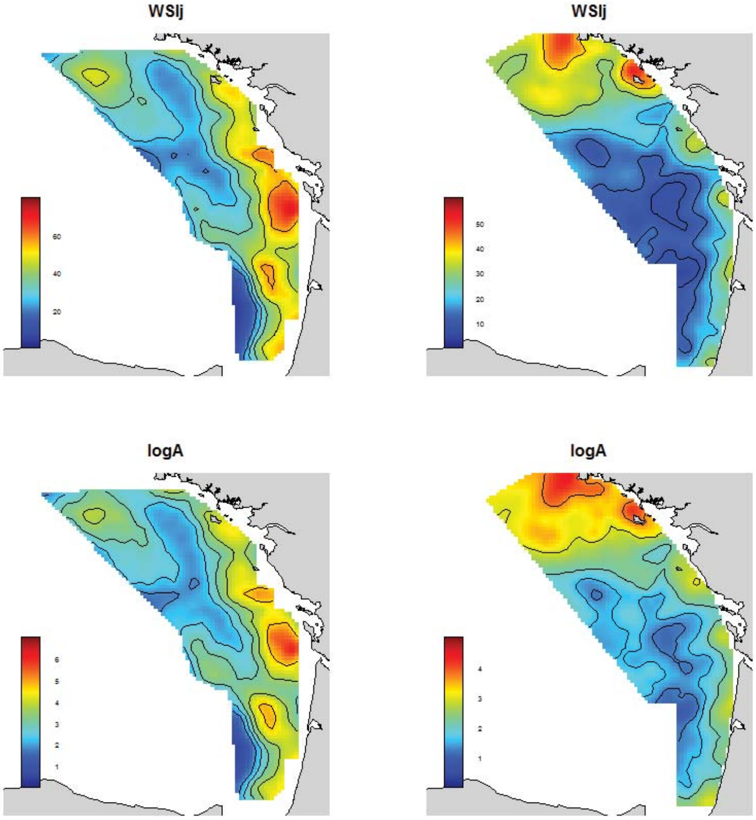

Fig.5 *WSI maps (up) and log abundance maps (down), left ROMER, right PELGAS*

to the one displayed by the simple sum of log abundances (Fig 5 c&d), indicating that SSIi has in fact a negligible effect on the impact assessment according to the original framework.

DISCUSSION

Usefulness of the refined framework

The approach originally developed by Garthe & Hüppop (2004) to assess the potential impact of offshore wind farm on seabird populations has several interesting aspects. The clear identification of species-specific risk factors and the method for scaling them is undoubtedly useful to synthesize quantitative and qualitative ecological information for impact assessment. It allows identifying which species is submitted to which risk, it is a catalyst for expert meeting groups and is a major methodological tool to reach a consensus between scientists and managers. However, the way this information was later on integrated and combined with survey data was not optimal. Some important piece of information was lost on the way, as for example the collision risk, and the original mathematical formulation would in fact result in taking decision based only on summed log-abundance patterns, instead of accounting for the additional information provided by the thorough documentation of all the risk factors and the computation of the SSIi.

The refined framework solves these issues. By explicitly separating primary risk factor and aggravation factors, by treating each risk separately, by integrating them sequentially first at the species level and then at the population level, and finally by explicitly separating community composition from abundances, we provides to manager all the pieces of information they would need to take informed decision, without implicitly masking some component of the impact. Because the framework is clearly mathematically defined, because all the assumptions are stated and written, we provides to scientist a transparent and tractable method for impact assessment. The method can be easily modified, would additional information appear on the way, concerning for example the way risk factors interact, or the relative importance of collision over disturbance. One important point is that we do not attempt to synthesize the information into one single map. Rather, we try to disentangle the different component of the information to present it in an integrated way to the manager. This is a very important aspect of communication between scientists and managers. While scientists usually try to identify all the aspects of a problem, managers seek simple answers and synthetic responses. This is one reason for the proliferation of indicator-based approaches (ref, ref, ref, ref). Our case study illustrate well that indeed, complex information related to the spatial distribution of 30 seabird species can be synthesized in a few set of maps showing different information. However, information reduction has to be carefully designed and firmly theoretically grounded. Reducing complex problems up to a single scale or a big formula may result in an un-tractable mixing of information that either becomes difficult to interpret or strongly under-estimates some crucial aspects of the problem.

Wind farm impact assessment in the Bay of Biscay

Our study allows to clearly localize areas of high and low expected impact on seabird for the establishment of offshore wind farm, as well as a qualitative assessment of the kind of impact to be expected. However, the reader should be aware that the quality of such an evaluation depends on the quality of the data. We have no doubt that ROMER and PELGAS surveys provided state-of-the-art data on seabird populations. However, these surveys have spatio-temporal limitations that need to be clearly stated. First, both surveys aimed the continental shelf and therefore, they do not document very well the coastal community, which is the reason why we do not map abundance or vulnerabilities near the coast. Second, the timing of the survey also limit the interpretation of our result. The ROMER survey focused on the wintering period, which is the period during which the seabird population is the highest in the Bay of Biscay (Certain 2007). PELGAS surveys offer the spring perspective, when already some of the main seabird taxa present in winter have left to reproduce, Auks for example. Therefore, we stress that the maps presented in this study can only serve for risk assessment during these time period, but that further surveys should be carried out and analysed to provide a better picture of the impact for the whole year round.

Potential for generality

The refined approach proposed in this paper has a much larger potential of application than the restricted scope of offshore wind farm impact assessment. The adaptation of the work of Leinster & Cobbold (2012) that we introduce (eq. 6) can in fact be applied to estimate the vulnerability of any kind of community to any kind of impact, provided that a measure of the species-specific vulnerability to that impact, such as vi, is available, and that community data have been collected. The application of eq. 6 is then straightforward. The fields within which this approach could be applied are numerous, all the more that the method provides an explicit link between an identified impact and a biodiversity-related metric at the community level. However, the method through which species-specific vulnerability can be measured is most likely case-study dependent, and should be each time carefully defined.

REFERENCES

Bellier, E., Certain, G., Planque, B., Monestiez, B., Bretagnolle, V., 2010. Modelling habitat selection at multiple scales with multivariate geostatistics: an application to seabirds in open sea. Oikos 119, 988–999.

Bellier, E., Monestiez, P., Certain, G., Chadoeuf, J., Bretagnolle, V., 2012. Decomposing the heterogeneity of species distributions into multiple scales: a hierarchical framework for large-scale count surveys. Ecography 35, 839–854.

Bretagnolle, V., Certain, G., Houte, S., Métais, M., 2004. Distribution maps and minimum abundance estimates for wintering auks in the Bay of Biscay , based on aerial surveys. Aquatic Living Resources 353–360.

Certain, G., Bellier, E., Planque, B., Bretagnolle, V., 2007. Characterising the temporal variability of the spatial distribution of animals: an application to seabirds at sea. Ecography 30, 695–708.

Certain, G., Bretagnolle, V., 2008. Monitoring seabirds population in marine ecosystem: The use of strip-transect aerial surveys. Remote Sensing of Environment 112, 3314–3322.

Certain, G., Skarpaas, O., Bjerke, J., Framstad, E., Lindholm, M., Nilsen, J.-E., Norderhaug, A., Oug, E., Pedersen, H.-C., Schartau, A.-K., van der Meeren, G.., Aslaksen, I., Engen, S., Garnasjordet, P.-A., Kvaloy, P., Lillegard, M., Yoccoz, N., Nybo, S., 2011. The Nature Index: A general framework for synthesizing knowledge on the state of biodiversity. PloS one 6, e18930.

Chadœuf, J., Certain, G., Bellier, E., Bar-Hen, a., Couteron, P., Monestiez, P., Bretagnolle, V., 2011. Estimating inter-group interaction radius for point processes with nested spatial structures. Computational Statistics & Data Analysis 55, 627–640.

Cressie, N.A.C., 1993. Statistics for spatial data. J. Wiley.

Drewitt, A., Langston, R.W., 2006. Assessing the impacts of wind farms on birds. Ibis 148, 29–42.

Exo, K.M., Hüppop, O., Garthe, S., 2003. Birds and offshore wind farms: a hot topic in marine ecology. Wader Study Group Bulletin 100, 50–53.

Garthe, S., Hüppop, O., 2004. Scaling possible adverse effects of marine wind farms on seabirds: developing and applying a vulnerabiblity index. Journal of Applied Ecology 41, 724–734.

Hill, M.O., 1973. Diversity and Evenness : A Unifying Notation and Its Consequences 54, 427–432.

Leinster, T., Cobbold, C., 2012. Measuring diversity: the importance of species similarity. Ecology 93, 477–489.

Masden, E. a, Haydon, D.T., Fox, A.D., Furness, R.W., 2010. Barriers to movement: Modelling energetic costs of avoiding marine wind farms amongst breeding seabirds. Marine Pollution Bulletin 60, 1085–91.

Pebesma, E.J., Wesseling, C.G., 1998. Gstat: A program for geostatistical modelling, prediction and simulation. Computers & Geosciences 24, 17–31.

Punt, M.J., Groeneveld, R. a., van Ierland, E.C., Stel, J.H., 2009. Spatial planning of offshore wind farms: A windfall to marine environmental protection? Ecological Economics 69, 93–103.

Tuomisto, H., 2010. A diversity of beta diversities: straightening up a concept gone awry. Part 1. Defining beta diversity as a function of alpha and gamma diversity. Ecography 33, 2–22.

RESUM

L'Índex de Sensibilitat als parcs eòlics (WSI) de les aus marines és una eina feta en el context dels sectors alemanys del mar Bàltic i del Nord. Va ser creat amb la finalitat de proporcionar una eina de decisió a escales grans per a l'avaluació ambiental estratègica de l'energia eòlica marina. A continuació, es mostra com es pot millorar l'índex d'una Avaluació Ambiental Estratègica (AAE) per al desenvolupament d'energia eòlica marina en un context de gran escala. El WSI integra la informació basada en les densitats d'aus marines a la zona d'estudi amb un Índex de Sensibilitat Espècies (SSI) als parcs eòlics. Aquest índex es calcula tenint en compte nou factors, que es deriven dels atributs de les espècies que semblen definir la sensibilitat de l'ocell amb els parcs eòlics. Es van dur a terme censos des de barca. Després de calcular el SSI per a cada espècie que es troba en l'àrea d'estudi, s'aplica l'índex als mapes de densitats locals obtinguts mitjançant els censos des de vaixell. Per prendre una decisió interessa treballar amb el mínim nombre de mapes que sintetitzin completament la realitat ecològica d'un àrea. Aquest index disposa d'aquesta característica integradora i el fa especialment interessant en l'avaluació ambiental estratègica d'un àrea determinada. Atès que hi ha una manca d'informació per conèixer l'abast de l'impacte real dels parcs eòlics en alta mar, es recomana aquest índex com un mètode molt útil en la majoria d'estudis d'avaluació ambiental estratègica mentre no es desenvolupi una eina millor per aquest tipus d'avaluació.

JOURNAL REFERENCE

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Wind farm Sensitivity Index for seabirds - Assessing offshore wind energy development on the coasts of the Iberian Peninsula

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ABSTRACT

The Windfarm Sensitivity Index (WSI) for seabirds is a tool made in the context of the German sectors at the Baltic and North Seas. It was created in order to provide a tool of decision for a broad-scale Strategic Environmental Assessment for offshore wind energy. Here, it is showed how this Index can improve an Strategic Environmental Assessment (SEA) for offshore wind energy development in a large scale context. The WSI integrates the information based on the seabirds densities in the study area with a Species Sensitivity Index (SSI) to windfarms. Such Index was calculated taking into account nine factors, which derive from the species attributes that seem to define the bird sensitivity to windfarms. Boat surveys were carried out. After calculating the SSI for each species found in our surveys, we applied it to the maps of local densities obtained by means of boat surveys. When making a decision, the fewer number of fully explicative maps are always desirable. This integrative characteristic of the index makes it especially interesting in the environmental assessment of a proposed offshore windfarm. Since there is a lack of information to know the extent of the real impact of offshore windfarms, we recommend this index as a very useful method in most SEA until the moment we will have a better tool for assessment.

INTRODUCTION

Renewable energies are viewed as an environmental benign alternative to the energy production based on fossil fuels (Inger et al., 2009). The potential of the marine environment, and particularly the offshore wind energy development, has received high attention in the last years. Europe has become the world leader in offshore wind power with a total of 1,371 offshore turbines totalling 3,812.6 MW spread across 53 wind farms in 10 countries by the end of 2011 (EWEA, 2012). At a global scale, the shift to renewable energies is widely accepted as a step to mitigate the effects of anthropogenically induced climate change (King, 2004; Rosenzweig et al., 2008). At the local scale, however, the environmental impacts of wind energy development must be carefully considered. Indeed, the European legislation requires Strategic Environmental Assessments (SEAs) of national wind energy plans impacts on wildlife (Directive 2001/42/EC).

Among the different topics that SEAs must address, wind farms and birds interactions are an issue of great concern (Garthe and Hüppop, 2004; Fox et al., 2006).Seabirds are susceptible to multiple anthropogenic impacts in their migratory routes and foraging grounds(Anderson et al., 2003; Hüppop et al., 2006; Louzao et al., 2006). In the case of offshore wind farms these potential impacts are direct mortality through collision, barrier effects and foraging habitat loss (Fox et al., 2006).

Beyond conservation concerns, seabirds have also become useful indicators to evaluate the potential effects of human activities in marine ecosystems (Piatt et al., 2007). Compared to other marine species, seabirds are highly visible species with specific legal protection frameworks and comprehensive long-term studies of their distribution at sea. Thus, seabirds seem a suitable indicator of the marine environment, and have become one of the keystones of the decision-making process for the selection of optimal areas for national offshore wind development.

The Wind Farm Sensitivity Index (WSI) was the first index that used seabirds to assess at large scale the suitability or unsuitability of a sea region for the construction of offshore wind farms. The index, developed by Garthe and Hüppop (2004), takes into account two crucial points in the evaluation of future impacts. First, it takes into account seabirds abundances and areas of high density which is the information commonly used to inform SEAs and EIAs on offshore wind energy. Second, the abundance of seabirds is corrected by a specific value (SSI, Species Sensitivity Index) that quantifies the sensitivity of each seabird species to the presence of an offshore wind farm or its construction. This way the presence of few individuals of flagship species can be accounted as well as the massive presence of common species, with no conservation concern but relevant because their numbers.

Despite the WSI relevance as a practical assessment tool, the peer-reviewed papers that have actually used it are adaptations of the index to evaluate other types of hazards (e.g. Noguera et al., 2010; Stelzenmüller et al., 2010; Sonntag et al., 2012) and so far there is no paper showing its application in a different geographic area and only some reports(e.g. Leopold and Dijkman, 2010; Christensen-dalsgaard et al., 2011).In this paper we present the WSI applied to the coasts of the Iberian Peninsula. The study area surrounds more than 7000 km and covers the continental coast of Portugal and Spain.

The Iberian Peninsula hosts the highest diversity of seabirds in Europe, mainly because its waters cover different biogeographical regions. Among these species some have their breeding stronghold in the Iberian Peninsula (e.g. the Balearic Shearwater Puffinus mauretanicus, Audouin's Gull Larus audouinii) and are flagspecies because their conservations status. Many of the species are listed in the Annex I of the European Community Birds Directive and are main targets of conservation projects (Ramirez et al., 2008; Arcos et al., 2009). So far, in the whole Iberian Peninsula there is only one experimental floating turbine installed in Portugal in June 2012. Therefore this is an opportunity to inform policy-makers and practitioners on how to design the optimal zonation to allow a rational offshore wind energy development respectful with the marine ecosystem.

The main aims of this paper are to: (i) suggest some changes in the factors used to calculate the SSI in order to make the Index more general and applicable to other biogegraphic areas in Europe and other continents; (ii) provide new SSI values to expand the original table by including the diversity of Atlantic and Mediterranean species detected in the area and (iii) make recommendations for the future development of offshore wind energy in the Iberian coasts with full awareness of ecological impacts.

METHOD

Study area &vessel-based data

At sea seabird surveys were conducted in different vessel expeditions coordinated by the Spanish Ornithological Society (SEO/Birdlife) and the Portuguese Society for the study of birds (SPEA). The surveys were carried out from 1999 to 2011 covering the Spanish and Portuguese continental coasts (up to 100 nautical miles offshore) (Fig 1a). The total surveyed area covered more than 25000 Km2 (Table 1). Seabird counts followed standardized strip-transect techniques (Tasker et al., 1984) adapted to the study area characteristics (Louzao et al., 2006). The observers covered a 300m strip transect band

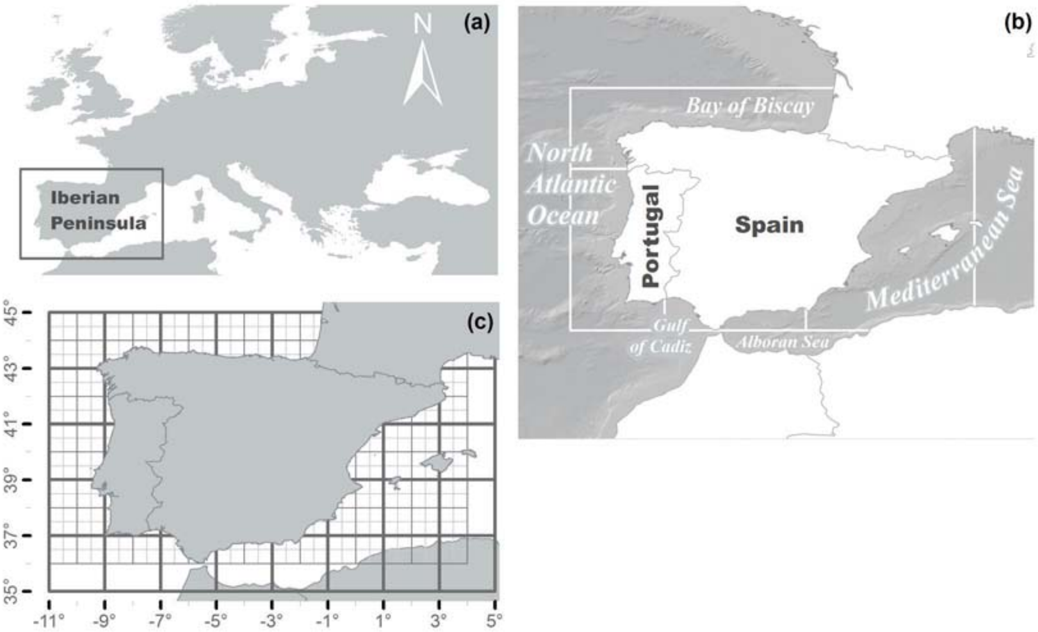


Fig.3 a) *Situation Map.* **b)** *Study regions marked by solid lines; I: Spanish North Atlantic Ocean, II: Portuguese North Atlantic Ocean, III: Gulf of Cadiz and Alboran Sea and IV: Mediterranean Sea.* **b)** *Grid location*

Table 1 *Total surveyed area (Km2) by year and month*

		Wintering		Breeding				Post Breeding				Wintering		
Year	Organization	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total Km^2
1999	SEO	-	-	-	-	701	-	-	-	375	-	-	-	1076
2000	SEO	-	-	-	-	691	-	-	-	-	-	-	-	691
2002	SEO	-	-	-	-	732	-	-	-	-	-	-	-	732
2003	SEO	-	-	-	-	-	-	-	-	-	-	830	-	830
2004	SEO + SPEA	-	-	-	-	-	-	-	-	-	-	618	78	696
2005	SEO + SPEA	531	415	7	488	339	260	1083	74	21	38	1033	26	4314
2006	SEO + SPEA	-	14	43	523	1139	28	7	71	529	157	1194	3	3708
2007	SEO + SPEA	6	263	66	690	1148	10	156	-	-	109	464	773	3686
2008	SEO + SPEA	160	310	12	535	229	77	219	4	-	500	528	-	2573
2009	SEO + SPEA	-	-	90	505	-	1194	-	-	-	-	-	350	2139
2010	SEO + SPEA	-	-	-	1168	116	9	1931	166	863	-	214	-	4467
2011	SEO	-	-	-	-	-	-	243	-	48	-	-	-	290
	Total Km^2	697	1001	218	3910	5095	1578	3640	315	1836	804	4881	1229	25203

at each side of the boat when visibility and wind conditions were adequate. All seabirds observed within the survey transect were recorded and summed into 10 minutes survey bins.

The study area had a latitudinal span of 10 decimal degrees (from 35° N to 45° N) and a longitudinal span of 15 decimal degrees (from 11° W to 4° E). It was divided in a regular grid at four different scales (2°, 1°, 0.5° and 0.25°) (Fig 1c). Four main regions have been considered for local discussion according to their oceanographic features and political boundaries: the Bay of Biscay and the Galician Atlantic coast (Region I), the Portuguese Atlantic coast (Region II), the Gulf of Cádiz and Alboran Sea (Region III) and the Mediterranean Sea (Region IV) (Fig 1b).

Species Sensitivity Index Calculation

The WSI is derived from distributional data of seabirds' counts at sea and a Species-specific Sensitivity index (SSI). SSI evaluates the species' vulnerability to wind farms using nine factors: flight manoeuvrability, flight altitude, percentage of time flying, nocturnal flight activity, sensitivity towards disturbance by ship and helicopter traffic, flexibility in habitat use, biogeographical population size, adult survival rate, and conservation status (Garthe and Hüppop, 2004). Each factor was scored on a 5-point scale where 1 indicated low vulnerability and 5 indicated high vulnerability. Following Garthe and Hüppop methodology, when no empirical data was available, the factors (5 out of 9) were given a subjective value partially based on bibliography -when available- and the authors experience on the species. These scores where then submitted for assessment to 10 independent experts with at-sea experience. After the independent evaluation, species scores where revised. When close species with similar characteristics had different values the experts where consulted again, and the values were corrected if a consensus was reached. The nine factors are shortly described below, with emphasis on those factors with changes from its original definition. A more detailed description of the factors can be found in the original paper (Garthe and Hüppop, 2004).

(F1) Flight manoeuvrability

This factor takes into account the flight ability of a species to avoid collision with wind farms at sea. Species were classified from very high flight manoeuvrability (score 1) to low flight manoeuvrability (score 5), and this classification wassent for evaluationtothe experts.

(F2) Flight altitude

This factor estimates how often a species flies within the range of the blades of the turbines. The altitudes were classified as follows: 1, 0-5m; 2, 5-10m; 3, 10-20m; 4, 20-50m and 5, 50-100m. The original factor was based on real data from flight altitude assessments. Since this information was not available in our surveys the experts were asked for the most frequent altitude class.

(F3) Percentage of time flying

This factor aims to assess how much time a species present in a wind farm area is susceptible to collision. This factor was obtained from the behavioural data collected during the surveys. Species were scored: 1, if 0-20% of the individuals were flying; 2, 21-40%; 3, 41-60%; 4, 61-80% and 5, 81-100%. After collecting and ranking behavioural data for each species, some discrepancies where found between similar species. These differences were artifacts of data related to differential detection. In these cases the scores where equalized for the group using the better sampled species.

(F4) Nocturnal flight activity

Nocturnal flight activity was classified from hardly any flight activity at night (score 1) to high flight activity at night (score 5) and evaluated by the experts.

(F5) Disturbance by ship and helicopter traffic

The ship and helicopter traffic during construction and maintenance of wind farms is expected to have some disturbance effect on species provoking escape, avoidance or fleeing behaviours. There is almost no information on the issue; hence the factor was evaluated subjectively from hardly any behavioural response (score 1) to strong behavioural reactions (score 5).

(F6) Flexibility in habitat use

This factor takes into account the habitat preferences of species. Those species occupying large sea areas and no specific habitat preferences (e.g. gulls) are expected to be less sensitive to offshore wind farms than those species relying on specific habitat features (e.g. sea ducks feeding on banks on shallow grounds). Therefore species were classified from very flexible in habitat use (score 1) to reliant on specific habitat characteristics (score 5) and again evaluated by the experts.

(F7) Biogeographical population size

Population sizes were obtained for each species from Birdlife publications (BirdLife International, 2004, 2012). Species were scored: 1 for populations exceeding 3 million individuals; 2 for 1-3 million individuals; 3 for 500000-1 million individuals; 4 for 100000-500000 individuals and 5 for less than 100000 individuals.

(F8) Adult survival rate

Additional mortality due to collisions is likely to affect species with high annual survival rates rather than species with low survival rates. The factor was classified as follows: 1, ≤ 0.75; 2, >0.75-0.80; 3, >0.80-0.85; 4, > 0.85-0.90; 5, > 0.90. The survival rates were obtained from Garthe and Hüppop (2004), Schreiber and Burger (2002) and Álvarez and Velando (2007).When the rate was not available the values from closely related species were taken.

(F9) Conservation status

The original factor reflected the European threat and conservation status using part of the SPEC (Species of European Concern) categories. For a more general index with applicability to any part of the world we used the IUCN conservation criteria. According to their conservation status species where scored: 1 for Least Concern; 2, Near threatened; 3, Vulnerable; 4, Endangered and 5, Critically Endangered.

With all the species scores the final SSI value was calculated for each species according to the following equation:

$$SSI = \frac{(F_1 + F_2 + F_3 + F_4)}{4} \times \frac{(F_5 + F_6)}{2} \times \frac{(F_7 + F_8 + F_9)}{3}$$

Distributional data and Wind farm Sensitivity Index maps

Once the SSI score is calculated for each species, the index is applied to the distributional data and transformed into the WSI. The count data of all the years was pooled and summarized in four temporal scenarios: the whole year, breeding season (March to June), post-breeding season (July to October) and wintering season (November to February). For each temporal scenario and spatial scale the WSI values were calculated. For each species the density per grid cell was obtained by dividing the sum of individuals by the total surveyed area in the cell. With this information the WSI value of the cell was:

$$WSI = \sum\nolimits_{species} \left(\ln(density_{species} + 1) \times SSI_{species} \right)$$

For each map, the WSI values were ranked and plotted in a colour gradient where each colour indicates a particular percentile. For the local discussion, the WSI values at the 0.25° scale were split in the four defined regions, ranked and plotted with independent colour gradient scales.

RESULTS

A total of 41 different species were counted in the surveys. The most abundant species were the Northern gannet *Morus bassanus* (32807 individuals), the Yellow-legged gull *Larus michahellis* (20449 individuals) and the Balearic shearwater *Puffinus mauretanicus* (12621 individuals). The species showed a wide range of sensitivity index values (Table 2). The Audouin's gull *Larus audouinii* and the Balearic shearwater were the species with the highest sensitivity while the Black-headed gull *Larus ridibundus*, the Little gull *Larus minutus* and the Atlantic puffin *Fratercula arctica* ranked the lowest.

For the whole year, the areas of highest vulnerability, i.e. with WSI values over the 50 percentile, were in the Portuguese Atlantic coast, the northern half of the Mediterranean region and the Gulf of Cadiz (Fig.2, 2° grid). At lower scales, where coastal cells could be differentiated from offshore cells, the values near the coast were generally higher than those further offshore, except for the Mediterranean region (Fig.2, 1° and 0.5° grid). The areas with lowest vulnerability were two: the limit area between the Alboran Sea and the Mediterranean region and the Eastern part of the Bay of Biscay.

Regarding the temporal evolution of vulnerable areas, the coastal middle part of the Portuguese region (between 38-42° N) and the Gulf of Cadiz remained vulnerable through the three different periods, whereas other areas increased their vulnerabi-

Table 2 *Total surveyed area (Km2) by year and month*

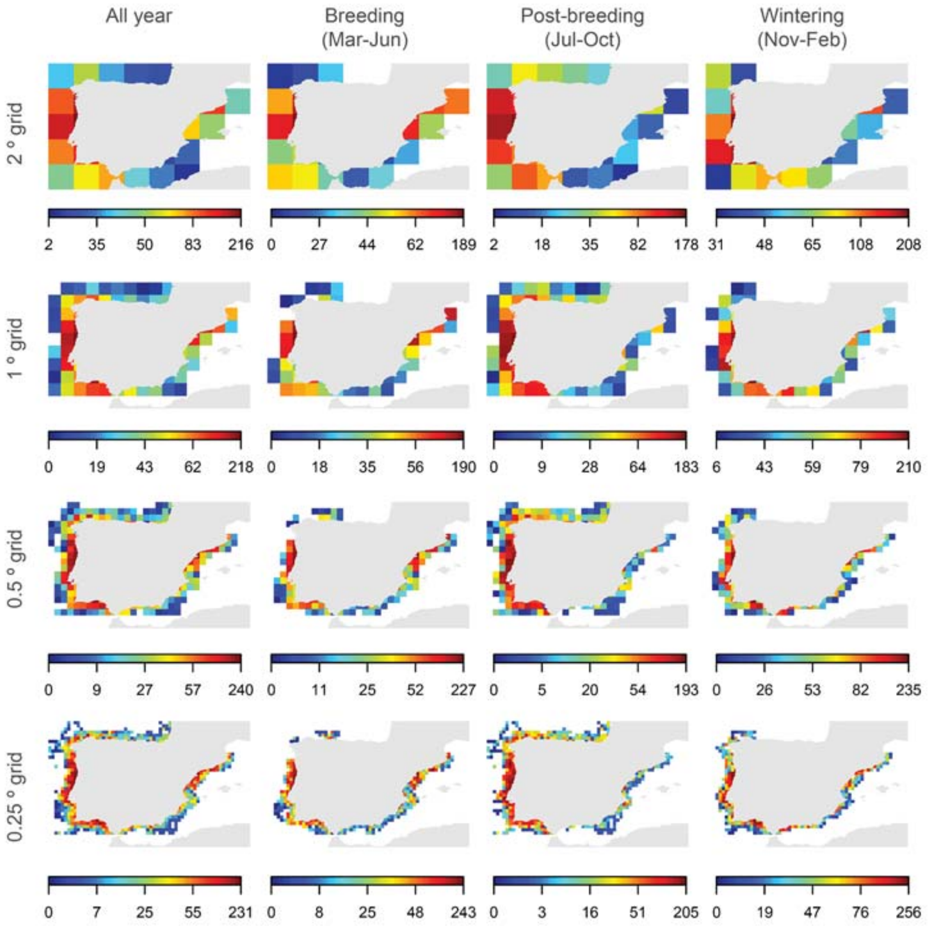
Group	Common name	Scientific name	F1	F2	F3	F4	F5	F6	F7	F8	F9	SSI
Procellariiformes	Cory's Shearwater	*Calonectris diomedea*	3	3	4	3	2	2	3	5	1	19.5
	Northern Fulmar	*Fulmarus glacialis*	3	2	4	3	2	2	1	5	1	14.0
	European Storm-petrel	*Hydrobates pelagicus*	2	1	4	5	2	3	2	5	1	20.0
	Wilson's Storm-petrel	*Oceanites oceanicus*	2	1	4	5	2	3	1	5	1	17.5
	Madeiran Storm-petrel	*Oceanodroma castro*	2	1	4	5	3	3	5	5	1	33.0
	Leach's Storm-petrel	*Oceanodroma leucorhoa*	2	1	4	5	3	3	1	5	1	21.0
	Great Shearwater	*Puffinus gravis*	3	3	4	3	2	2	1	5	1	15.2
	Sooty Shearwater	*Puffinus griseus*	3	3	4	3	2	2	1	5	2	17.3
	Balearic Shearwater	*Puffinus mauretanicus*	3	2	4	2	2	3	5	5	5	34.4
	Manx Shearwater	*Puffinus puffinus*	3	3	5	3	2	3	2	5	1	23.3
	Yelkouan Shearwater	*Puffinus yelkouan*	3	2	4	2	2	3	5	5	3	29.8
Cormorants	European Shag	*Phalacrocorax aristotelis*	3	2	2	1	4	4	4	3	1	21.3
	Great Cormorant	*Phalacrocorax carbo*	4	4	5	1	4	2	2	3	1	21.0
Gannets	Northern Gannet	*Morus bassanus*	3	4	4	2	3	2	3	5	1	24.4
Skuas	Great Skua	*Catharacta skua*	2	4	4	2	2	2	5	4	1	20.0
	Long-tailed Jaeger	*Stercorarius Longicaudus*	2	4	4	2	3	2	3	3	1	17.5
	Parasitic Jaeger	*Stercorarius parasiticus*	2	4	4	2	3	2	3	3	1	17.5
	Pomarine Jaeger	*Stercorarius pomarinus*	2	4	4	3	2	3	4	4	1	24.4
Gulls	Herring Gull	*Larus argentatus*	2	4	4	3	3	1	2	5	1	17.3
	Audouin's Gull	*Larus audouinii*	2	4	4	5	2	3	5	5	2	37.5
	Lesser Black-backed Gull	*Larus fuscus*	2	4	4	4	2	2	3	5	1	21.0
	Slender-billed Gull	*Larus genei*	2	3	4	2	4	4	4	4	1	33.0
	Mediterranean Gull	*Larus melanocephalus*	2	4	4	2	2	3	3	4	1	20.0
	Yellow-legged Gull	*Larus michahellis*	2	4	4	3	3	1	2	5	1	17.3
	Little Gull	*Larus minutus*	1	2	4	2	2	3	4	2	1	13.1
	Black-headed Gull	*Larus ridibundus*	2	3	4	2	2	2	1	3	1	9.2
	Sabine's Gull	*Larus sabini*	2	3	4	3	3	3	4	4	1	27.0
	Black-legged Kittiwake	*Rissa tridactyla*	1	3	4	3	3	3	1	4	1	16.5
Terns	Black Tern	*Chlidonias niger*	2	2	5	2	3	3	4	4	1	24.8
	Little Tern	*Sterna albifrons*	2	2	5	2	3	3	4	4	1	24.8
	Common Tern	*Sterna hirundo*	2	3	5	3	3	3	2	4	1	22.8
	Gull-billed Tern	*Sterna nilotica*	2	3	5	1	3	3	5	4	1	27.5
	Arctic Tern	*Sterna paradisaea*	2	3	5	2	3	3	3	4	1	24.0
	Sandwich Tern Sterna	*Sterna sandvicensis*	2	3	5	2	3	3	3	4	1	24.0
Auks	Razorbill	*Alca torda*	4	1	3	1	3	3	2	5	1	18.0
	Atlantic Puffin	*Fratercula arctica*	4	1	3	1	2	3	1	5	1	13.1
	Common Guillemot	*Uria aalge*	4	1	3	1	3	4	1	4	1	15.8
Seaducks	Common Scoter	*Melanitta nigra*	3	2	5	3	5	4	2	2	1	24.4
Waders	Red Phalarope	*Phalaropus fulicarius*	2	1	3	3	4	2	2	3	1	13.5

lity in particular periods of the year (Fig.2, 1° grid). In the breeding months (March to June), the northern half of the Mediterranean region (between 39-42°N) showed high levels of vulnerability. In the post-breeding months (July to October), the Mediterranean region showed moderate to low vulnerability levels while vulnerability was increased in the coast of the western part of region I. The eastern part of region I, corresponding to the Bay of Biscay, was only surveyed during the post-breeding months hence no information is available for the breeding and wintering season.

The regional vulnerability map (Fig. 3), highlights the vulnerability of the Galician coast (western area of region I) as well as the Northern area of region II, the Gulf of Cadiz (region III) and the Northern half of the Mediterranean region (region IV). These vulnerable areas fit well with the already defined Marine Important Bird Areas (IBAs), except in the Northern part of Portugal, which shows the highest WSI values of the Iberian Peninsula and has no marine protected area. All over the peninsular coasts, even in areas with low vulnerability, there are punctual areas near the coast with high vulnerability.

The WSI values and their increment at each percentile were similar across grid scales and temporal scenarios (Fig. 4). Nevertheless, at broad scales the index values were slightly higher. Between temporal scenarios the increment of the WSI values was almost identical except in the wintering period when the index values were higher.

Fig. 2 *Wind farm Sensitivity Index values in time and scale*



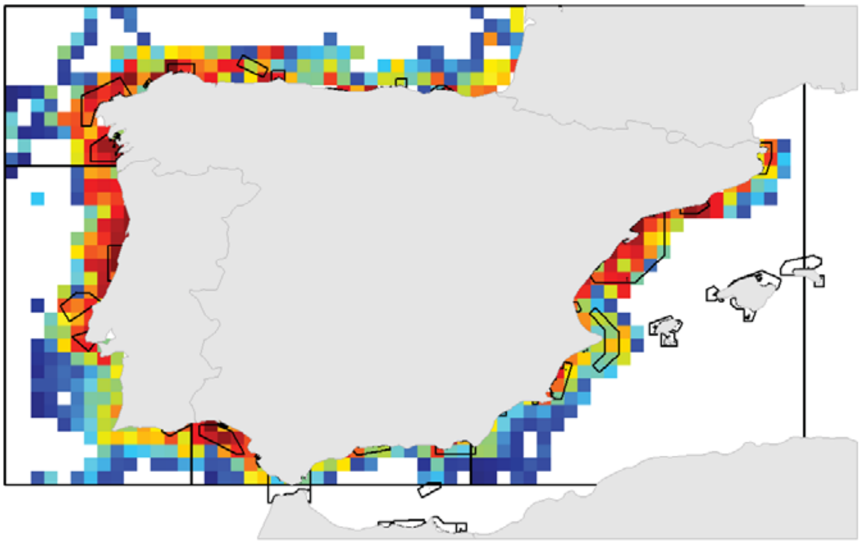


Fig. 3 *WSI values by regions (scale 0.25° and all year). Marine IBA overlayed in each region. Each region has its own colour scale.*

DISCUSSION

SSI and WSI modifications

The technological advances in remote sensing has fostered the study of seabird movements at sea (Ropert-Coudert and Wilson, 2005; Louzao et al., 2009; Christel et al., 2012). Nevertheless there is still a lack of information about seabirds' behaviour in offshore areas and how this behaviour can be affected by the presence of offshore wind farms (Desholm and Kahlert, 2005; Perrow et al., 2011). Thus, the SSI methodology that merges real data with expert-based scores is a useful tool to evaluate species sensitivity to wind farms when no quantitative data is available.

The scoring method of the SSI is flexible and can easily be adapted to data availability and circumstances of a particular study, however, we found necessary to make an important change in the conservation status factor. We suggest using the IUCN Red list categories by default instead of the SPEC (Species of European Concern) categories

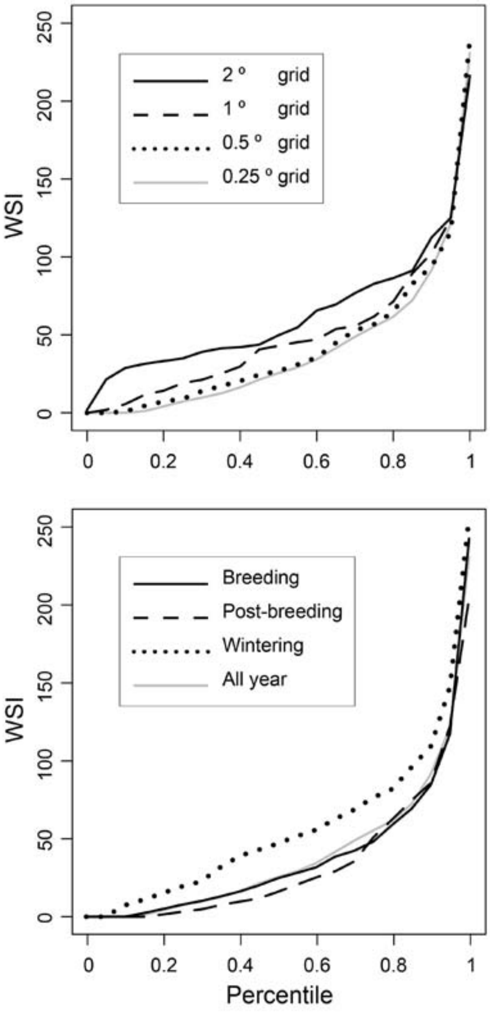


Fig. 4 *WSI values by scale and season*

and only choose other conservation classifications if all the study species fall in the same category which was the case of Garthe and Hüppop's study. The SPEC categories are not as widely recognized as the IUCN categories. Moreover, this classification criterion is only useful in a European framework and offshore wind energy development and any recommended tool should aim to use international standards.

Concerning the WSI graphical output, the original methodology suggested three levels of categorization where cells with WSI values over the 60thpercentile were defined as 'concern' areas and those over the 80th percentile were defined as 'major concern' areas. In the reports that have applied this methodology these thresholds have been retained. In the papers that have adapted the methodology a similar system has been used except in one case (Stelzenmüller et al., 2010) where the results were rescaled to a 1 to 10 qualitative scale. Although three levels seems an intuitive output for decision-making, the selection of the threshold percentiles that divides concern areas from low/no concern areas is subjective. Besides, the 60th and 80th percentiles might not be the adequate cut-off values for all biogeographic regions. Instead of generalizing these values, we propose to plot the WSI vulnerability maps ranking the cells from the lowest WSI value to the highest. Plotting a ranking allows a comparative analysis between high and low concern areas, retains the percentile information and at the same time avoids the subjective definition of a threshold.

Vulnerability map and Offshore Wind Energy planning

The 2° grid is not adequate, the 1° and 0.50 better. 1° is the size region of the Spanish marine zonation for offshore wind farm SEA. The optimal development areas (South Mediterranean region in Spain, south in Portugal, Bay of Biscay but there is poor sampling compared to other areas!)

Always better further offshore than near the coast. Avoid the Mediterranean areas with high vulnerability are mainly related to breeding period. The most important species (highest SSI) are Larus audouinii and Puffinus mauretanicus.

Galician coast has high vulnerability due to the post-breeding migration. Portuguese coast high vulnerability due to high numbers of Puffinus mauretanicus and high concentrations at sea of Melanitta nigra

Limitations to the technique: new methods like satellite tracking are complementing the information of offshore distribution of flagship species. With every study new patterns are discovered and areas that at seem less important according to boat surveys may emerge as areas of intensive use outside the hours covered with surveys. That's the case of the Cape of Naos (Spanish coast in front of Ibiza) which is an area of extensive use of Puffinus mauretanicus and based on the boat surveys data would be of low vulnerability.

CONCLUSION

This SSI table could be of major interest in the future if offshore wind energy is to be developed in Spain, Portugal or any other country with similar diversity of seabirds. Some changes suggested to make the index more internationally applicable. Better avoid the percentile threshold and use ranking instead.

With this study we demonstrate the utility of the WSI as a comparative tool at large scale but also applicable with at smaller scales for detail. Applying the WSI to the Iberian Coasts we show how the index is useful to highlight development areas as well as priority areas for conservation regarding seabirds.

The resulting vulnerability maps show seabirds' spatial patterns as density maps do, but they also emphasize the presence of key species with higher sensitivity to wind farms. For policy makers and conservation practitioners the fewer number of fully explicative maps are always desirable for decision-making. This integrative characteristic of the index makes it especially interesting for the assessment of large areas.

REFERENCES

Anderson, D., Huyvaert, K., Wood, D., 2003. At-sea distribution of waved albatrosses and the Galapagos Marine Reserve. Biological Conservation 110, 367–373.

Arcos, J., Bécares, J., Rodríguez, B., Ruiz, A., 2009. Áreas Importantes para la Conservación de las Aves marinas en España, IBAS marinas book. LIFE04NAT/ES/000049-Sociedad Española de Ornitología (SEO/Birdlife), Madrid.

BirdLife International, 2004. Birds in Europe: population estimates, trends and conservation status, BirdLife Conservation Series n°12. BirdLife International, Cambridge.

BirdLife International, 2012. IUCN Red List for birds, httpwwwbirdlifeorgactionsciencespeciesglobalspeciesprogrammeredlisthtml. BirdLife International. Available: http://www.birdlife.org (accessed 17 July 2012), Cambridge.

Christel, I., Navarro, J., del Castillo, M., Cama, A., Ferrer, X., 2012. Foraging movements of Audouin's gull (Larus audouinii) in the Ebro Delta, NW Mediterranean: A preliminary satellite-tracking study. Estuarine, Coastal and Shelf Science 96, 257–261.

Christensen-dalsgaard, S., Lorentsen, S., Hanssen, F., Systad, G.H., 2011. Marine wind farms and seabirds. Updated screening of potential conflict areas on a national scale. NINA Report 616.

Desholm, M., Kahlert, J., 2005. Avian collision risk at an offshore wind farm. Biology Letters 1, 296–298.

EWEA, 2012. The European offshore wind industry key 2011 trends and statistics. European Wind Energy Association.

Fox, A.D., Desholm, M., Kahlert, J., Christensen, T.K., Petersen, I.K., 2006. Information needs to support environmental impact assessment of the effects of European marine offshore wind farms on birds. Ibis 148, 129–144.

Garthe, S., Hüppop, O., 2004. Scaling possible adverse effects of marine wind farms on seabirds: developing and applying a vulnerabiblity index. Journal of Applied Ecology 41, 724–734.

Hüppop, O., Dierschke, J., Exo, K.M., Fredrich, E., Hill, R., 2006. Bird migration studies and potential collision risk with offshore wind turbines. Ibis 148, 90–109.

Inger, R., Attrill, M.J., Bearhop, S., Broderick, A.C., James Grecian, W., Hodgson, D.J., Mills, C., Sheehan, E., Votier, S.C., Witt, M.J., Godley, B.J., 2009. Marine renewable energy: potential benefits to biodiversity? An urgent call for research. Journal of Applied Ecology 46, 1145–1153.

King, D.A., 2004. Climate Change Science : Adapt, Mitigate, or Ignore? Science 303, 176–177.

Leopold, M.F.L., Dijkman, E.M., 2010. Offshore wind farms and seabirds in the Dutch Sector of the North Sea: What are the best and the worst locations for future develpment? Institute for Marine Resources & Ecosystem Studies.

Louzao, M., Bécares, J., Rodríguez, B., Hyrenbach, K., Ruiz, a, Arcos, J., 2009. Combining vessel-based surveys and tracking data to identify key marine areas for seabirds. Marine Ecology Progress Series 391, 183–197.

Louzao, M., Hyrenbach, K.D., Arcos, J., Abelló, P., de Sola, L.G., Oro, D., 2006. Oceanographic habitat of an endangered Mediterranean procellariiform: Implications for marine protected areas. Ecological Applications 16, 1683–1695.

Noguera, J.C., Pérez, I., Mínguez, E., 2010. Impact of terrestrial wind farms on diurnal raptors: developing a spatial vulnerability index and potential vulnerability maps. Ardeola 57, 41–53.

Perrow, M.R., Gilroy, J.J., Skeate, E.R., Tomlinson, M.L., 2011. Effects of the construction of Scroby Sands offshore wind farm on the prey base of Little tern Sternula albifrons at its most important UK colony. Marine Pollution Bulletin 62, 1661–70.

Piatt, J.F., Harding, A., Shultz, M., Speckman, S., van Pelt, T., Drew, G.S., Kettle, A., 2007. Seabirds as indicators of marine food supplies: Cairns revisited. Marine Ecology Progress Series 352, 221–234.

Ramirez, I., Geraldes, P., Meirinho, A., Amorim, P., Paiva, V.H., 2008. Important Areas for Seabirds in Portugal. Project LIFE04NAT/PT/000213. Sociedade Portuguesa Para o Estudo das Aves, Lisboa.

Ropert-Coudert, Y., Wilson, R.P., 2005. Trends and perspectives in animal-attached remote sensing. Frontiers in Ecology and the Environment 3, 437–444.

Rosenzweig, C., Karoly, D., Vicarelli, M., Neofotis, P., Wu, Q., Casassa, G., Menzel, A., Root, T.L., Estrella, N., Seguin, B., Tryjanowski, P., Liu, C., Rawlins, S., Imeson, A., 2008. Attributing physical and biological impacts to anthropogenic climate change. Nature 453, 353–357.

Schreiber, E.A., Burger, J. (Eds.), 2002. Biology of Marine Birds. CRC Press, Boca Raton.

Sonntag, N., Schwemmer, H., Fock, H.O., Bellebaum, J., Garthe, S., 2012. Seabirds, set-nets, and conservation management: assessment of conflict potential and vulnerability of birds to bycatch in gillnets. ICES Journal of Marine Science 69, 578–589.

Stelzenmüller, V., Ellis, J.R., Rogers, S.I., 2010. Towards a spatially explicit risk assessment for marine management: Assessing the vulnerability of fish to aggregate extraction. Biological Conservation 143, 230–238.

Tasker, M.L., Jones, P.H., Dixon, T. m, Blake, B.F., 1984. Counting seabirds at sea from ships: a review of methods employed and a suggestion for a standardized approach. The Auk 101, 567–577.

Álvarez, D., Velando, A., 2007. El cormorán moñudo en España. Población en 2006-2007 y método de censo. Madrid.

RESUM

El desenvolupament de l'energia eòlica marina ha fomentat el debat sobre l'impacte potencial d'aquestes infraestructures sobre les aus marines. En aquest context apareix la necessitat de trobar indicadors que determinin l'efecte i extensió d'aquests impactes. La majoria d'Estudis d'Impacte Ambiental (EIA) presenten mapes de distribució i densitat d'aus, però molt pocs intenten representar de manera explícita els impactes potencials en l'espai i el temps. Mitjançant la relació entre mitjana i variància descrita per Taylor (Taylor 's power law) i models lineals mixtos es pot modelar la variabilitat espai-temporal dels patrons de distribució de les aus marines. Els models resultants descriuen el grau d'agregació de les aus al mar el que permet diferenciar zones de transició d'àrees d'alimentació. Aquesta distinció, al seu torn, es pot utilitzar per definir zones amb un alt risc de col · lisió i zones de potencial pèrdua d'hàbitat en el cas de construir un parc eòlic marí. Amb el Delta de l'Ebre com a cas d'estudi il · lustrem la utilitat d'aquest mètode i comentem els avantatges dels mapes d'impacte potencial respecte als mapes d'abundància.

JOURNAL REFERENCE

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Seabird aggregative patterns: a new tool for offshore wind energy risk assessment

3

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ABSTRACT

The emerging development of offshore wind energy has raised public concern over its impact on seabird communities. There is a need for an adequate methodology to determine its potential impacts on seabirds. Environmental Impact Assessments (EIA) are mostly relying on a succession of plain density maps without integrated interpretation of seabird spatio-temporal variability. Using Taylor's power law coupled with mixed effect models, the spatio-temporal variability of species' distributions can be synthesized in a measure of the aggregation levels of individuals over time and space. Applying the method to a seabird aerial survey in the Ebro Delta, NW Mediterranean Sea, we were able to make an explicit distinction between transitional and feeding areas to define and map the potential impacts of an offshore wind farm project. We use the Ebro Delta study case to discuss the advantages of potential impacts maps over density maps, as well as to illustrate how these potential impact maps can be applied to inform on concern levels, optimal EIA design and monitoring in the assessment of local offshore wind energy projects.

INTRODUCTION

Studies on marine top predators are today considered as a key component of marine ecosystem management (Boyd et al., 2006). Within top predators, seabirds are good indicators of ecosystem health (Cairns, 1987; Mallory et al., 2006; Nettleship and Duffy, 1993) and are useful indicators to evaluate potential effects of human activities at sea. Most seabirds are flagship species for the public (Fox et al., 2006) and have clear protection criteria collected in protection directives like the Birds directive (79/409/EEC) and Habitats directive (92/43/EEC) in Europe. Their distribution and abundance are usually provided as key information to support the establishment of marine protected areas, to implement fisheries' management measures (Boyd et al., 2006), to assess the impact of environmental disasters such as oil spills (Bretagnolle et al., 2004; Moreno, 2011) or to monitor the impact of oil and gas platforms at sea (Wiese et al., 2001).

In the last years, offshore wind energy has emerged as a priority field in many European countries to meet Europe's 2020 agenda that promotes renewable energies to mitigate the effects of climate change; hence offshore wind farms will likely experience an important increase in the near future. However, in the field of marine management there is a growing concern on the development of offshore wind energy and its potential impacts on coastal seabird populations, mainly because of possible collisions with windmills (Fox et al., 2006). On a large scale, countries might develop "Strategic Environmental Assessments" (SEA) to plan their offshore wind farms network in a way that minimizes their ecological impact on the coastal environment (Directive 2001/42/EC). At a local scale, each wind farm project requires an Environmental Impact Assessment (EIA) of its potential impact in the marine environment, including the risk imposed on avian populations (Bright et al., 2008; Masden et al., 2010).

The potential impacts of offshore wind farms on seabird communities are complex. Fox et al. (2006) provided a conceptual classification of these impacts, distinguishing between (1) avoidance, (2) modification of the physical habitats, and (3) direct mortality trough collision. Most EIA guidelines suggest radar studies to assess collision risk in strongly migratory areas (Desholm et al., 2006; Fox et al., 2006; Kunz et al., 2007) and density maps as a proxy to loss of foraging habitats by avoidance and physical habitat modification (Camphuysen et al., 2004; Fox and Petersen, 2006). However, density maps do not provide a full understanding of the underlying behavioral patterns related to their movements. Seabirds often present dynamic and complex spatial patterns at sea which are far from being understood. When foraging, many species of seabirds are usually characterized by an important aggregative behavior (Buckley, 1997; Grünbaum and Veit, 2003), with birds forming flocks of hundreds of individuals. On the contrary, a lower aggregative behavior is expected in transitional areas solely used as flight paths between feeding areas and their resting or breeding areas. While density maps focus on high concentrations of seabirds as potential risk areas, we propose the explicit distinction between transitional and foraging areas as a key step to better predict and classify the risk of wind farm establishment on seabird populations. In transitional areas, the main risk will be direct collision and mortality (Desholm and Kahlert, 2005; Hüppop et al., 2006). In foraging areas, the risk of direct collisions is increased and potentially associated with a displacement from their preferred feeding areas, resulting in habitat loss (Masden et al., 2010; Perrow et al., 2011).

In 2004, the proposal of an offshore wind farm project in front of the Ebro Delta (North-Western Mediterranean, Fig. 1a) emphasized the necessity for adequate indicators to determine the extent and effect of potential impacts on its seabird community. Here, we use the slope of the Taylor's power law as a measure of the aggregative patterns of seabirds to identify transitional and feeding areas, and map the risk accordingly. The slope of the Taylor's power law (Taylor, 1961; Taylor and Woiwod, 1982) provides a convenient measure of the aggregation levels of animals (see Kendal, 2004 for a review). It has already been used in a spatio-temporal context with seabirds (Certain et al., 2007) and has proved to be useful to describe the temporal variability associated to the spatial distribution of seabirds at multiple scales. Here, using the Ebro Delta as a case study, we first show how to take into account the aggregative properties of seabird distributions together with abundance maps. Second, we point the advantages of this method as an integrative tool to summarize in few maps the spatial and temporal variability of the potential impacts of offshore wind farms. Finally, we discuss how the resulting potential impacts maps provide a frame to inform on EIA design and monitoring in the context of an offshore wind farm proposal.

METHOD

Study area & survey method

The Ebro Delta (NW Mediterranean, Fig. 1) is a very productive area because of a permanent upwelling, result of the sudden broadening of the shelf (up to 70km) in combination with the influence of the Liguro-Provençal-Catalán front and nutrients carried by the Ebro river runoff (Arcos et al., 2001; Palomera, 1992). This high productivity supports an important fishing fleet with a high trawling activity (Arcos et al., 2001; Louzao et al., 2006; Palomera, 1992) which in turn has been pointed as a key resource for seabirds (Arcos, 2001; Arcos et al., 2008). However, the trawling activity is regulated with temporal moratoria in the area. Fishing moratoria affects the northern area (B1-2 and B14-16, Fig. 1) in May and June and the southern area (B3-B13) during July and August, and influences the distribution of

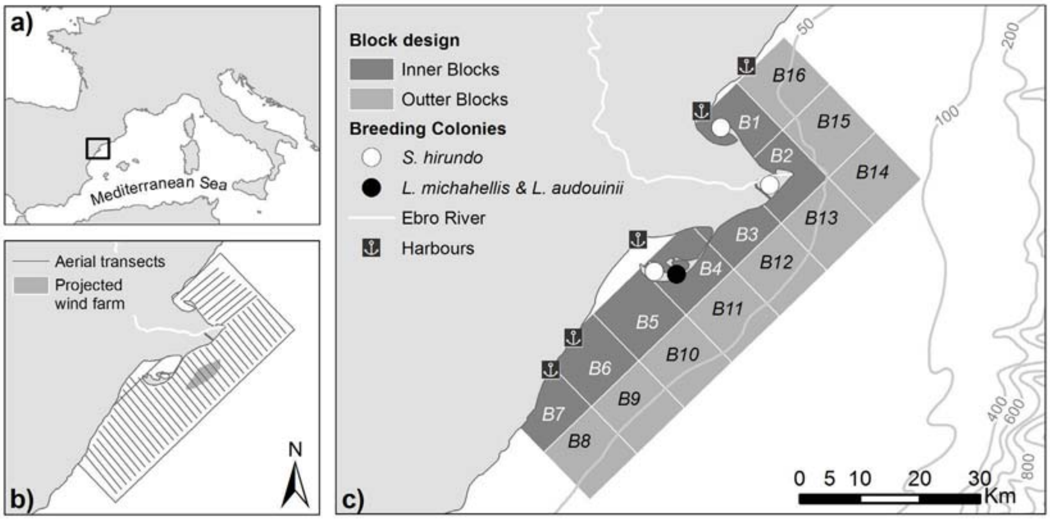


Fig.1 **(a)** *Situation map.* **(b)** *Survey design of aerial transects and projected offshore wind farm location.* **(c)** *Block design of the study area showing inner and outer classification and block Id. The main breeding colonies location, harbours and the Ebro River are shown.*

some species.

Seven monthly aerial surveys were carried out from March 2005 to September 2005 on the continental shelf around the Ebro Delta (40.7° N, 0.75° E). The surveys covered a total area of 1435 km^2 from L'Ametlla de Mar harbour (24 km North; 40.86° N, 0.8° E) to Peñíscola (51 km South; 40.35° N, 0.4° E) (Fig. 1). The entire shelf area can be covered in a single day using this approach, and availability biases due to attraction and avoidance movements of seabirds were minimized. In this study, we used the standard seabird aerial survey methodology described by Noer et al. (2000).

The survey area was covered by 45 transects systematically arranged in parallel lines running perpendicular to the coast, to follow the dominant sea depth gradient, and flown at 2 km intervals. During the surveys, two observers, one at each side of the aircraft, covered 1 km strip at each side. The surveys were conducted from a twin-engine aircraft, Partenavia P68, and the aircraft GPS was used for navigation along the transect tracks. The cruising speed was set at c. 100 knots (185 km/h) with respect to the air speed and average flying height was 300 feet (100 m). Along the transects, all observed bird flocks were recorded with a voice recorder, stating information on species (or the lowest taxonomical level determinable), number of individuals, behaviour (e.g. flying, flushing, sitting on water, feeding on trawler discards), age whenever possible, transect strip, date and time. The presence of trawlers was also recorded. These recordings were geo-referenced later with the transect track information provided by a GPS and a Turbo Pascal application (Ib Krag Petersen pers. com.). In the moments of maximum glare or any other adverse light situation, the counting was interrupted. Since counts results are highly sensitive to meteorology, no surveys were conducted when Beaufort Sea state was greater than one.

Table 1. *Descriptive data*

	Mar	Apr	May	Jun	Jul	Aug	Sep
Surveyed length (km)	2099	3337	3575	3977	2583	2745	2539
Seastate (Douglas scale)[a]	0	1	0	0	1	0	1
Cloud Coverage[b]	B	A	A	A	B	A	B
Larus michahellis							
Total counted individuals	3803	1894	683	226	360	666	1801
Total sightings	462	241	295	119	78	110	393
Larus audouinii							
Total counted individuals	269	385	183	562	165	176	244
Total sightings	68	111	130	239	83	62	39
Sterna hirundo							
Total counted individuals	18	92	464	628	2568	4324	2635
Total sightings	4	39	225	281	293	262	179

[a] *Based on Douglas scale*

[b] *A) Sunny, without cloud coverage; B) Partially covered with clouds*

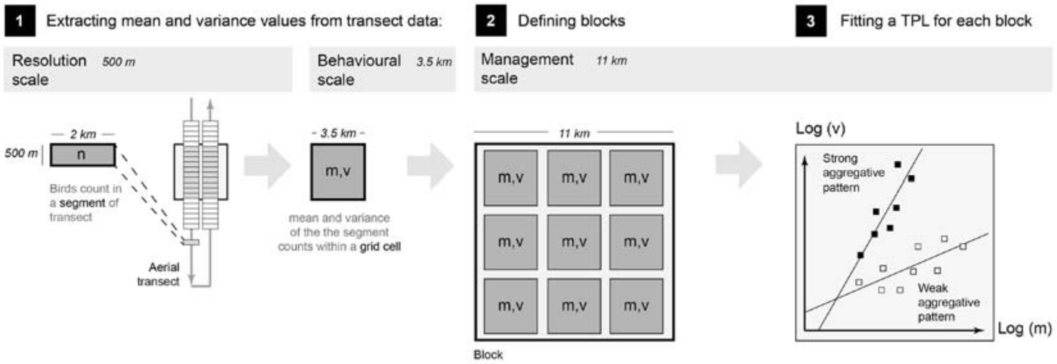


Fig.2 *Data preparation and selected scales.*

Study model

We focused the study on the three most abundant seabird species in the area, the Yellow-legged Gull (*Larus michahellis*), the Audouin's Gull (*Larus auouinii*) and the Common Tern (*Sterna hirundo*). These three species represented the 93% of all detected individuals at sea. Moreover, they can be used as model species for their different foraging and feeding strategies. The Yellow-legged Gull is a scavenger that makes extensive use of trawler's discards (Oro et al., 1995), and their foraging habits are strongly influenced by trawlers predictability (Cama et al., 2012). The Audouin's Gull is an opportunist species that exploits small pelagic fish (Oro, 1998 and references therein), but also makes use of trawler's discards and terrestrial food sources (Christel et al., 2012; Navarro et al., 2010; Oro and Ruiz, 1997). The Common Tern, conversely, only preys actively on shoals of small pelagic fish (Cramp and Simmons, 2004).

The Yellow-legged Gull population in the Ebro Delta is sedentary. The species breeds from mid-March to April at the Punta de la Banya peninsula (Fig.1). Some individuals of Audouin's Gull are in the area all over the year, but the main population is migratory as well as the Common Tern population. Both species are present in the area between March and September. The species arrive to the breeding grounds in March and April, being the peak of the breeding season between May and June, after which there is a variable post-fledging period with dispersive behaviour until they start their migration from late August to October (Cama, 2010). In 2005, the main colony for the Common Tern was in the north of the Ebro Delta with 3361 breeding pairs, and the main colony for the Yellow-legged and the Audouin's gulls was in the South of the Ebro Delta having 9850 and 13850 breeding pairs respectively (Fig. 1) (Cama, 2010).

Data preparation

First, transects were sliced into segments of 0.5 km length, each segment containing the number of birds counted for each species (Fig. 2). This length corresponds to the minimum scale at which the information could be located, according to the survey protocol (Noer et al., 2000). Second, mean and variance of bird abundance of the segments were computed within grid cells of 3.5 km wide. Only grid cells containing a minimum of 10 segments and at least two non-zero abundance values were included in the analyses. Finally, the study area was further divided in blocks of 11km. These 11km correspond to the scale at which management information is extracted, and was set as a trade-off for sample size. We searched the finer scale that had at least N=8 grid cells into each block so that to fit a power law and provide detailed information for management. At this selected block size, the pure coastal areas could be distinguished from areas located more offshore, where bird activity might differ in abundance and behavior.

Modeling of seabird aggregative pattern

Pioneered work of Taylor (1961) on the spatial and temporal variability of animal abundance provides an useful framework to study the spatio-temporal heterogeneity of a population within its habitat (Kendal, 2004; Kilpatrick and Ives, 2003; Taylor and Woiwod, 1980, 1982). Taylor's power law (referred as TPL from here onwards) states that the variance in abundance (V) is proportional to a power of the mean abundance (M):

$$V = b \times M^{a} \qquad \text{(Eq. 1)}$$

Which in the logarithmic scale becomes a linear regression, where a is the slope and log b is the intercept:

$$\log(V) = a \times \log(M) + \log(b) \qquad \text{(Eq. 2)}$$

In this context, the slope a is considered an aggregation index. If individuals are randomly distributed the slope equals 1, if individuals show some degree of aggregation the slope increases accordingly (Engen et al., 2008; Kendal, 2004; Kilpatrick and Ives, 2003). When calculated through space, the TPL slope can be interpreted as a measure of the strength of the aggregative response of organisms (Jiménez et al., 2001; Östman, 2002). Usually, slope values range between 1 and 2 when estimated in space (Engen et al., 2008; Kendal, 2004). When calculated through time, it can be used as an index of the temporal variability of the spatial distribution of organisms, highlighting recurrent and occasional presence areas (Certain et al., 2007).

Coupling Taylor's power law with linear mixed effect models (LME) allows the investigation of the spatio-temporal variability of TPL slope and consequently the variability of the aggregative patterns of organisms, avoiding confusing effects of changes in animal abundances (Certain et al., 2007).

The simplest model within this framework is:

$$\log(V_{bmsj}) = A \times \log(M_{bmsj}) + B + \varepsilon_{bmsj}$$
$$j = 1,\ldots,n_{bms} \qquad \text{(Eq. 3)}$$

Where the slope and the intercept are supposed constant through space (b), time (m) and the three species (s). However, the slope might vary according to one or several of these factors. The effect of these factors and all the possible combinations can be introduced in the model as a grouping factor with a random effect on the slope (Pinheiro and Bates, 2000). Starting from the simplest possible model (Eq. 3), different models were developed by the sequential addition of random effects on the slope. The most complete model could be written as the following:

$$\log(V_{bmsj}) = (A + a_{bms}) \times \log(M_{bmsj}) + B + \varepsilon_{bmsj}$$
$$j = 1,\ldots,n_{bms} \qquad \text{(Eq. 4)}$$

Where A is the fixed slope, a_{bms} is the random effect on the slope of block, month and species together, B is the fixed intercept, n_{bms} is the number of observations on a bms combination, and the ε_{bmsj} are independent

$N(0,\sigma^2)$ error terms. Forward stepwise model selection was applied. Each model was compared with the null model with a likelihood ratio test to check whether or not the inclusion of a new grouping factor was outperforming the previous one (Pinheiro and Bates, 2000). We retained the simplest model for which the inclusion of any new grouping factor did not result in a significant improvement of the model. All data processing and model developments were performed in R (R Development Core Team, 2008) with package lme4 (Bates et al., 2011).

To visualize the slope variation across months, the predicted slope values of the optimal mixed effect model were averaged by month (Fig. 3). Spatial variation of the slope was summarized for the breeding (March to June) and post-breeding (July to September) seasons, and was calculated with the average of the slope values for the corresponding months and plotted together with the density map of the number of individuals per month and block area (Fig. 4).

Behavioral interpretation of the aggregative pattern

We employed a generalized additive model (GAM) to test if the resulting slope values of the mixed effect model had any significant linear or non-linear correlation with the density of seabirds, the recorded behavior, the flock size or the presence of trawlers. For each block in a given month, we extracted the density of seabirds (total observed individuals divided by block area), mean and variance of the size of the observed flocks and the number of trawlers. The behavioral information recorded with the observations was classified in two categories: i) Flying, ii) fishing (on shoals or vessels) or sitting on water; and used to calculate the proportion of birds flying. The GAM analysis was carried out with the mgcv package (Wood, 2006) following a forward stepwise model selection based on the minimization of AIC and the analysis of deviance between models. The number of knots in the smooth functions was minimized to five to avoid overfitting.

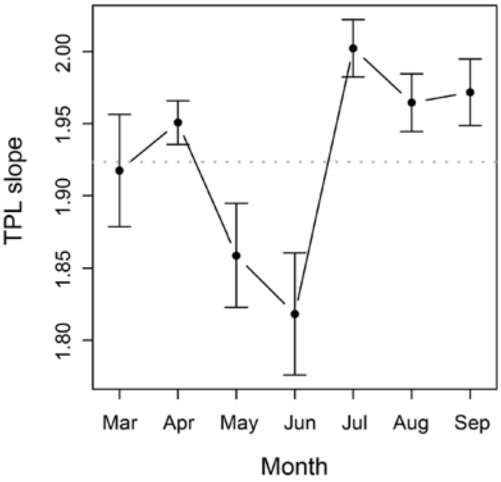


Fig.3 *Monthly average and standard error of TPL slope values for all blocks. The general mean slope for the full area and months is showed in grey dotted line.*

RESULTS

The result of the model-selection process for the mixed-effect model is presented in table 1. The optimal model retained block and month as the grouping factor with a random effect on the slope. Species had no significant effect on the slope. The fixed estimated slope for all the area was 1.923 and the predicted slope values ranged between 1.712 and 2.049.

The monthly evolution of Taylor's power law slope (Fig. 3) can be summarized in two sequences. First, from March to June (Mean±SE = 1.89±0.02), and especially in

Table 2. *Forward-selection procedure used to find the best set of predictors of Taylor's power law slope in the mixed effects model.*

Formula			AIC	Dev
Fixed Effect		Random Effect		
Var ~ Mean			1000.9	96.1
Var ~ Mean	+	B	955.2	80.9
Var ~ Mean	+	M	972.3	80.6
Var ~ Mean	+	S	1002.9	80.3
Var ~ Mean	**+**	**BM**	**904.8**	**81.7**
Var ~ Mean	+	BS	951.7	81.2
Var ~ Mean	+	MS	980.4	80.5
Var ~ Mean	+	BM + BS	906.6	81.7
Var ~ Mean	+	BM + MS	906.5	81.7
Var ~ Mean	+	BS + MS	934.7	81.5
Var ~ Mean	+	BMS	934.7	81.4

AIC is the Akaike information criterion, and Dev indicates the percentage of deviance explained by each model. Abbreviations for the formula terms are: Var, Logarithm of the variance (Dependent variable); Mean, logarithm of the mean; B, Block; M, month; S, Species. The model emboldened was selected as the optimal one.

May and June, slope values are low, suggesting low aggregation levels of the seabirds populations at sea. Then, TPL slope increases markedly from July to September (Mean±SE = 1.97±0.01), suggesting strong aggregations of seabirds at sea.

As two time periods (March-June and July-September) were clearly distinguished in the temporal evolution of the TPL slope in the area, spatial maps of TPL slope were drawn for these two periods (Fig. 4). From March to June, blocks near the colonies (B4-5, B10 and B1-2) had slope values under the average, suggesting a lower aggregative behavior, while areas in the outer blocks near the river mouth (B13-12) had higher slopes. From July to September, the slope values around the colonies increased switching to a more aggregated pattern (except B4). The outer blocks north and south to the river mouth (B11-16) retained and intensified their aggregated pattern. Southern blocks (B6-9) did not show any constant pattern in slope values between seasons.

According to the optimal selected GAM model (AIC= -135, deviance explained= 29.3%), the predicted slope values show a linear negative correlation with the proportion of flying birds (Estimate=-0.12826, p=0.006) and a non-linear correlation with the interaction of mean flock size and density (p=0.005, 3.734 estimated degrees of freedom). When factors were examined one by one, mean flock size was the main driver of slope changes (p<0.0001, 22.8% deviance explained) followed by proportion of flying birds (p<0.0001, 14.8%). Density had no significant effect (p=0.216, 1.73%).

Thus, an increase in the mean flock size in a block increases TPL slope, but blocks with high numbers of flying birds are more likely to have lower slopes. The main effect of density is on its interaction with flock size. For low density values, an increase on the mean flock size has a logarithmic increase in TPL slope. For densities greater than five individuals per km^2 the increase becomes linear (Fig. 5).

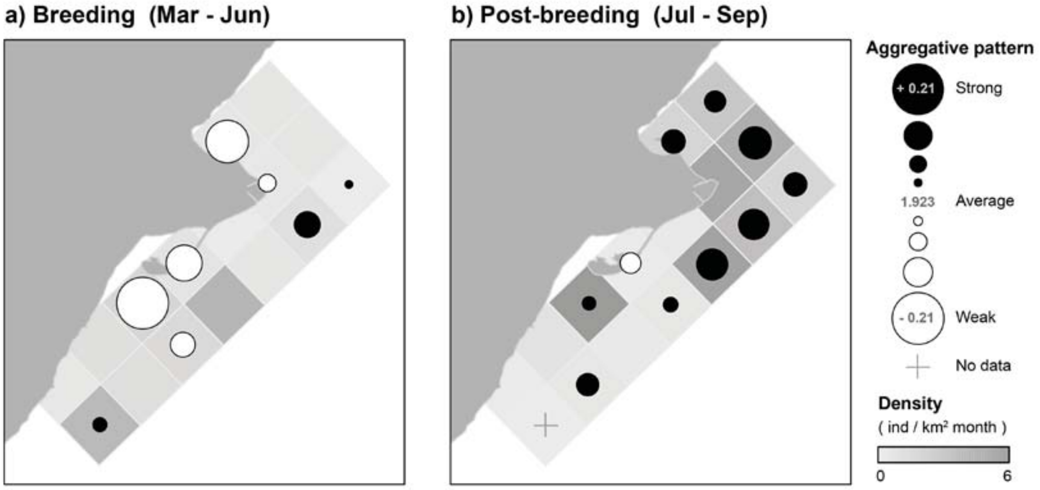


Fig.4 *Spatial structure of TPL slope values for the two seasons. Only values with intervals (Mean±SE) significantly different to the general average (1.923) are plotted. In the background density values (average ind/km2) are shown in grey scale.*

DISCUSSION

Aggregative patterns in time and space

The species of seabirds observed in the continental shelf were mainly gulls and terns in high numbers. Despite the selected species have different feeding sources and foraging behaviors, the species of both groups are central place foragers (Orians and Pearson, 1979) and use the presence of conspecifics as a cue to find food patches at sea (Paiva et al., 2007; Ward and Zahavi, 1973). This common behavior probably explains the similarity in the aggregative patterns between species.

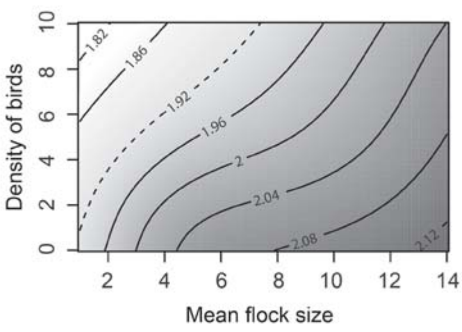
The temporal evolution of the aggregative patterns (Fig. 3) is strongly correlated to the life cycle of the species. The marked decrease in TPL slope in May and June coincides with the chick-rearing period, when most of the pairs of the three species already have chicks and perform short and frequent foraging trips resulting in low aggregation levels at sea. After the breeding season in June, TPL slope increases markedly as birds are freed from chick-rearing constraints, suggesting strong aggregations of birds at sea. The seasonal pattern of TPL slope suggests that these two periods can be used to summarize and highlight the main behavioural and aggregative patterns of the population. Indeed, seasonal scenarios are easier to communicate than a detailed sequence of monthly representations and are useful, for instance, to recommend mitigation measures. Regarding the spatial structure of the aggregative patterns at both seasons (Fig. 4), the differences between the blocks near the breeding colonies, the blocks in the outer side of the study area and the southernmost blocks can be explained respectively by the vital cycle requirements, the feeding sources distribution, and trawling moratoria influence.

The area near the colonies has low aggregative levels during the breeding season. At this moment of the year the species perform frequent and shorter foraging trips than the rest of the year (Paiva et al., 2007). This results in a constant transit of individuals flying from and to the colony minimizing the time spent foraging. Once the chick-rearing period ends, the aggregative patterns around colonies increase; although the area might still be a highly transited area, it is probably combined with some groups of individuals feeding or spending time resting in these areas.

The aggregated pattern of the outer blocks is likely to be driven by the presence of feeding sources. Previous studies in the area have reported the extensive use of trawlers discards by gulls (Arcos et al., 2001; Cama et al., 2012; Navarro et al., 2010; Pedrocchi et al., 2002) and the diet preferences of common terns (Arcos et al., 2002; Hernández-Matías, 2003), but no work so far had characterized the spatio-temporal distribution of their aggregation at sea. The aggregation in the outer blocks south of the river mouth is due to a higher presence of trawlers and big flocks of birds associated (see trawlers' distribution in Cama et al., 2012). The area north of the river mouth is a highly productive area due to the Ebro river runoff (Palomera, 1992; Sierra et al., 2002) characterized by the abundance of clupeoids (mainly sardine *Sardine pilchardus* and snchovy *Engraulis encrasicholus*). The presence of fish shoals and the proximity of the common tern colony make this area an optimal feeding ground for the Common tern (Hernández-Matías, 2003). Besides, the presence of trawlers makes the northern area attractive for Audouin's gull and Yellow-legged gull during the trawling moratoria in the southern blocks (Cama, 2010).

The southern area has two main harbours (Vinarós and Benicarló) and an important daily discarding activity from 15 to 16 h which attracts large flocks of seabirds (Cama, 2010). However, southernmost blocks show a less consistent pattern between seasons and none differentiates significantly from the average aggregation level. This is most likely an effect of trawling moratoria on seabirds' presence in the area.

a) 20% of birds flying



b) 80% of birds flying

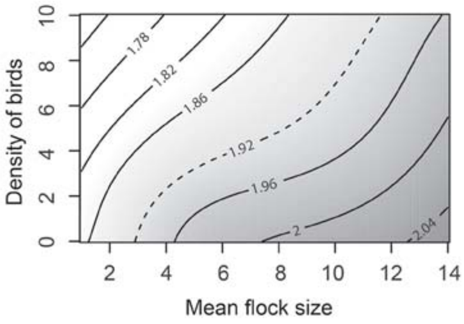


Fig.5 *Contour plots of the predicted TPL slope according to density of birds and mean flock size,* **(a)** *for a fixed 20% of birds flying and* **(b)** *for an 80%. The average TPL slope for the whole study area indicated with dashed line. Results based on the semi-parametric GAM model of TPL slope with a linear estimator for proportion of birds flying and a smooth term for the interaction of mean flock size and density.*

Behavioral interpretation of the aggregative pattern

Taylor's power law slope is widely accepted as an aggregation index, which is corroborated by its correlation with mean flock size.

The GAM analysis allows us to be more precise in the interpretation of TPL slope in the case of seabird populations in the Ebro Delta. Low TPL slopes are related to areas with high percentages of birds flying, usually individually or forming small flocks (Fig. 5b). Hence, areas with weak aggregative patterns can be considered transitional or flight path areas. High TPL slopes are found in areas with high percentages of birds feeding and –to a lesser extent– sitting on water, mainly forming big flocks (Fig. 5a).

This indicates that areas with strong aggregative patterns are mostly feeding areas

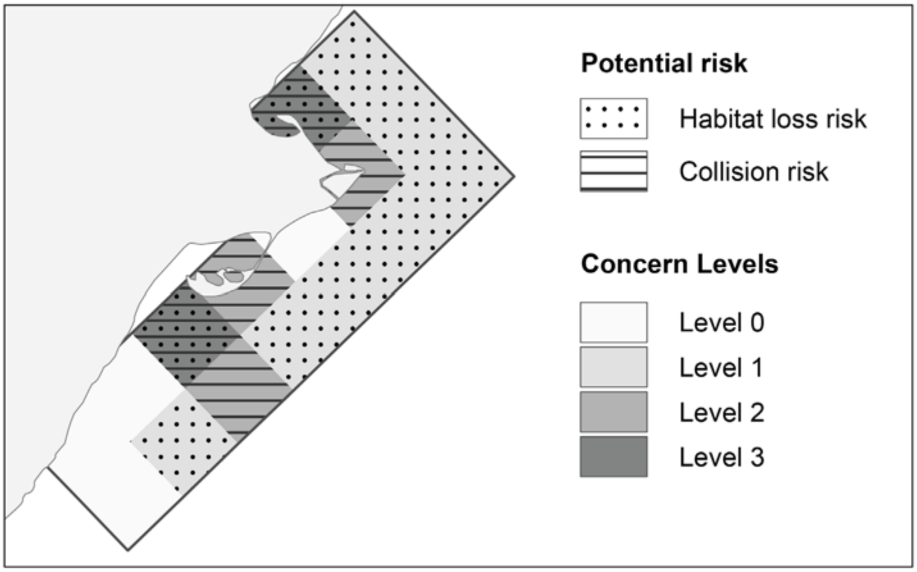


Fig.6 *Potential risk map (according to transitional and feeding areas) and concern levels for offshore wind farm placement.*

where the presence of birds' aggregations is driven by the availability of punctual feeding opportunities (fish shoals near the surface or discarding trawlers in this study case).

Density maps and Potential impact maps

Density maps define areas with high numbers of birds but they do not provide information on the dynamic and complex spatial patterns of seabirds. At the Ebro Delta, if a key protection area had to be selected according to density maps (Fig. 4 background) the northern area would be the one of highest concern, mainly based on high abundances during the post-breeding season. Despite the intense pattern of movements near the colonies, during the breeding period the pattern would be masked by low density values. Since the aggregative pattern is a reflection of behavior, it has a direct application in the demarcation of areas of high potential risk for seabirds. Areas revealed as main travelling areas are highly susceptible to collision threat. Areas pointed as foraging areas will eventually have big groups of seabirds feeding. The presence of offshore turbines in these areas would result in habitat loss threat for species with a strong avoidance response or an increased collision risk for the species that venture between the wind turbines (Fig. 6).

This is particularly true for the assessment of areas of known importance for breeding populations. However, we suggest the necessity of applying this methodology to flyway corridors or areas with a different composition of species (e.g. plunge-divers like gannets or surface-divers like sea ducks) to investigate any possible difference in the interpretation of the potential risks associated to the observed aggregative patterns. Nevertheless, to consider the information on the second order properties of species' distributions (i.e. social aggregation) provides further information to managers in terms of potential impacts of offshore wind farms than solely focusing on the first order properties (i.e. density).

Concern levels and monitoring protocol

The placement of an offshore wind farm is a management decision that takes into account many socio-economic and environmental factors. To facilitate the inclusion of seabirds in the decision-making process we propose a ranking of the areas according to their potential impacts on the seabirds' populations. Based on recommendations in the available literature, each level of concern should be associated to a set of required types of surveys for the EIA and the compulsory monitoring for pre- and post-construction of accepted wind farms.

Collision mortality is often considered to be the most important hazard (Fox et al., 2006). Accordingly, those areas with both Collision and Habitat loss risk would have the higher concern level (L3); areas with collision risk would have the next concern level (L2) followed by areas with habitat loss risk (L1).

Any project placed in L0 areas would require the monitoring of the nearest colonies to obtain estimates of demographic and population sizes to assess how the populations respond to the offshore wind farm (Kunz et al., 2007).

Any project placed in L1 areas, besides the previous recommended monitoring, should in addition include distribution and habitat modeling (Drewitt and Langston, 2006; Fox and Petersen, 2006). Part of the modeling can be based on the already available aerial surveys. However, species with limited numbers or the very small species (e.g. storm petrels, shearwaters, alcids) are likely missed by aerial surveys and only found in boat surveys (Camphuysen et al., 2004). Hence, L1 areas with planned wind turbines should be assessed with detailed boat-based surveys. Modeling species presence and abundance is crucial to describe the factors driving the aggregative patterns which in turn can be used to evaluate barrier effects and the energetic costs of avoidance (Masden et al., 2010). In the Ebro Delta study case, the movement of trawlers through time likely changes the distribution patterns of gulls at sea as the birds follow them (e.g. Cama et al. 2012), hence an offshore wind farm would only change the distribution of this feeding source, having little impact on gulls. By contrast, the distribution of terns is driven by productivity and fish shoals availability (Paiva et al., 2007). If high numbers of the species were observed in the area selected for the wind farm, the expected habitat loss could have harmful effects on the population and should be taken into account in the EIA.

For any project placed in L2 areas, besides L1 and L0 monitoring, collision risk models should be calculated. This requires -depending on the particular case- point transect surveys of flight height and direction (Camphuysen et al., 2004), surveillance radars (vertical and/or horizontal) or infrared camera systems (Desholm et al., 2006). In these cases it is especially important an adequate monitoring in pre- and post-construction to evaluate predictions made in EIA, to allow adaptive management of the wind farm but also to quantify the cumulative impacts on migratory species (Fox et al., 2006).

Finally, for any project placed too near or inside Level 3 areas, satellite-tracking studies should be also carried out in order to assess quantitatively the intensity of tracks in the transitional areas and the recurrence of feeding areas. The selected species should be preferentially a flagship or keystone species, because of their conservation status or relevance in the community. In the Ebro Delta, for instance, the near threatened Audouin's gull (*Larus audouinii*) would be the target of the study since the colony holds 12000-13000 breeding pairs, ca. 65% of the world's total population of this species (Christel et al., 2012; Oro et al., 2009).

CONCLUSION

The design of EIA and monitoring surveys is not always an easy, straightforward decision. This is particularly true in areas where there is scarce knowledge on the distribution and abundance of the seabirds' community at sea. Unlike boat surveys, aerial surveys provide an extensive coverage in a short period of time that offers an image of the presence of seabirds at sea in a particular moment (Certain and Bretagnolle, 2008; Drewitt and Langston, 2006). This characteristic feature of aerial surveys can unveil distribution patterns that differ with the previous knowledge, which is often based in observations from land or ship-based surveys with more limited area coverage per survey than flights. Taylor's power law applied on aerial surveys provides a convenient analysis tool to ensure the optimal allocation in time and space of resources in order to obtain the most detailed knowledge for the EIA of future offshore wind farms on seabirds.

Although presented for a local scale, we think that this methodology would be very useful in the four steps of offshore wind energy development: the SEA of offshore wind energy development, the decision-making on wind farm projects placement, the EIA design and the monitoring planning of accepted projects. At a broader scale than the one presented here, this tool could be used in the marine spatial planning to select development regions that avoid the areas identified with potential impacts. Within a selected region, stating a clear monitoring protocol prior to placement would improve the decision-making process. To know the compulsory monitoring in a selected site might help to decide the optimal location of offshore wind farms minimizing not only the impact on the seabird community but also the future monitoring costs. Once the placement is decided, the same results could be used to inform the EIA.

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REFERENCES

Arcos, J., 2001. Foraging ecology of seabirds at sea: Significance of commercial fisheries in the NW Mediterranean. Thesis, University of Barcelona. <http://www.tdx.cat/handle/10803/775> (accessed 28.02.12)

Arcos, J., Louzao, M., Oro, D., 2008. Fisheries ecosystem impacts and management in the Mediterranean: seabirds point of view. American Fisheries Society Symposium 49, 1471-1479.

Arcos, J., Oro, D., Sol, D., 2001. Competition between the yellow-legged gull Larus cachinnans and Audouin's gull Larus audouinii associated with commercial fishing vessels: the influence of season and fishing fleet. Marine Biology 139, 807-816.

Arcos, J., Ruiz, X., Bearhop, S., Furness, R.W., 2002. Mercury levels in seabirds and their fish prey at the Ebro delta (NW Mediterranean): the role of trawler discards as source of contamination. Marine Ecology Progress Series 232, 281-290.

Bates, D.M., Maechler, M., Bolker, B., 2011. lme4: Linear mixed-effect models using S4 classes. R package version 0.999375-42.

Boyd, I.L., Wanless, S., Camphuysen, C.J., 2006. Top predators in marine ecosystems: their role in monitoring and management. Cambridge University Press.

Bretagnolle, V., Certain, G., Houte, S., Métais, M., 2004. Distribution maps and minimum abundance estimates for wintering auks in the Bay of Biscay, based on aerial surveys. Aquatic Living Resources 353-360.

Bright, J., Langston, R.W., Bullman, R., Evans, R., Gardner, S., Pearcehiggins, J., 2008. Map of bird sensitivities to wind farms in Scotland: A tool to aid planning and conservation. Biological Conservation 141, 2342-2356.

Buckley, N.J., 1997. Spatial-concentration effects and the importance of local enhancement in the evolution of colonial breeding in seabirds. American Naturalist 149, 1091–1112.

Cairns, D., 1987. Seabirds as indicators of marine food supplies. Biological Oceanography 5, 261-271.

Cama, A., 2010. Seabird distribution patterns and ecological factors driving larid presence at the Ebro delta shelf (NW Mediterranean). Thesis, University of Barcelona.

Cama, A., Abellana, R., Christel, I., Ferrer, X., Vieites, D.R., 2012. Living on predictability: modeling the density distribution of efficient foraging seabirds. Ecography. DOI: 10.1111/j.1600-0587.2011.06756.x.

Camphuysen, C.J., Fox, A.D., Leopold, M.F., Petersen, I.K., 2004. Towards standardised seabirds at sea census techniques in connection with environmental impact assessments for offshore wind farms in the U.K., Cowrie Report. London.

Certain, G., Bellier, E., Planque, B., Bretagnolle, V., 2007. Characterising the temporal variability of the spatial distribution of animals: an application to seabirds at sea. Ecography 30, 695-708.

Certain, G., Bretagnolle, V., 2008. Monitoring seabirds population in marine ecosystem: The use of strip-transect aerial surveys. Remote Sensing of Environment 112, 3314-3322.

Christel, I., Navarro, J., del Castillo, M., Cama, A., Ferrer, X., 2012. Foraging movements of Audouin's gull (Larus audouinii) in the Ebro Delta, NW Mediterranean: A preliminary satellite-tracking study. Estuarine, Coastal and Shelf Science 96, 257-261.

Cramp, S., Simmons, K.E.L., 2004. BWPi. Birds of the Western Paleartic on Interactive DVD-ROM. Birdguides, Oxford University Press, Oxford, U.K.

Desholm, M., Fox, A.D., Beasley, P.D.L., Kahlert, J., 2006. Remote techniques for counting and estimating the number of bird-wind turbine collisions at sea: a review. Ibis 148, 76-89.

Desholm, M., Kahlert, J., 2005. Avian collision risk at an offshore wind farm. Biology Letters 1, 296-298.

Drewitt, A., Langston, R.W., 2006. Assessing the impacts of wind farms on birds. Ibis 148, 29-42.

Engen, S., Lande, R., Sæther, B.-E., 2008. A general model for analyzing Taylor's apatial acaling laws. Ecology 89, 2612-2622.

Fox, A.D., Desholm, M., Kahlert, J., Christensen, T.K., Petersen, I.K., 2006. Information needs to support environmental impact assessment of the effects of European marine offshore wind farms on birds. Ibis 148, 129-144.

Fox, A.D., Petersen, I.K., 2006. Assessing the degree of habitat loss to marine birds from the development of offshore wind farms, in: Boere, G.C., Galbaraith, C.A., Stroud, D.A. (Eds.), Waterbirds Around the World. The Stationery Office, Edimburg, UK, pp. 801-804.

Grünbaum, D., Veit, R.R., 2003. Black-browed albatrosses foraging on antarctic krill: Density-dependence through local enhancement? Ecology 84, 3265–3275.

Hernández-Matías, A., 2003. An approach to coloniality in seabirds: The common tern Sterna hirundo as a case study. Thesis, University of Barcelona.

Hüppop, O., Dierschke, J., Exo, K.-michael, Fredrich, E., Hill, R., 2006. Bird migration studies and potential collision risk with offshore wind turbines. Ibis 148, 90-109.

Jiménez, J., Rossi, J.P., Lavelle, P., 2001. Spatial distribution of earthworms in acid-soil savannas of the eastern plains of Colombia. Applied Soil Ecology 17, 267-278.

Kendal, W.S., 2004. Taylor's ecological power law as a consequence of scale invariant exponential dispersion models. Ecological Complexity 1, 193–209.

Kilpatrick, A., Ives, A., 2003. Species interactions can explain Taylor's power law for ecological time series. Nature 422, 65–68.

Kunz, T.H., Arnett, E.B., Cooper, B.M., Erickson, W.P., Larkin, R.P., Mabee, T., Morrison, M.L., Strickland, M.D., Szewczak, J.M., 2007. Assessing Impacts of Wind-Energy Development on Nocturnally Active Birds and Bats: A Guidance Document. Journal of Wildlife Management 71, 2449-2486.

Louzao, M., Hyrenbach, K.D., Arcos, J., Abelló, P., de Sola, L.G., Oro, D., 2006. Oceanographic habitat of an endangered Mediterranean procellariiform: Implications for marine protected areas. Ecological Applications 16, 1683-1695.

Mallory, M.L., Gilchrist, H.G., Braune, B.M., Gaston, A.J., 2006. Marine birds as indicators of Arctic marine ecosystem health: linking the Northern Ecosystem Initiative to long-term studies. Environmental Monitoring and Assessment 113, 31-48.

Masden, E. a, Haydon, D.T., Fox, A.D., Furness, R.W., 2010. Barriers to movement: Modelling energetic costs of avoiding marine wind farms amongst breeding seabirds. Marine Pollution Bulletin 60, 1085-91.

Moreno, R., Jover, L., Diez, C., Sanpera, C., 2011. Seabird feathers as monitors of the levels and persistence of heavy metal pollution after the Prestige oil spill. Environmental Pollution 159, 2454-60.

Navarro, J., Oro, D., Bertolero, A., Genovart, M., Delgado, A., Forero, M.G., 2010. Age and sexual differences in the exploitation of two anthropogenic food resources for an opportunistic seabird. Marine Biology 157, 2453–2459.

Nettleship, D.N., Duffy, D.C., 1993. Seabird populations. Elsevier Applied Science, London.

Noer, H., Christensen, T.K., Clausager, I., Petersen, I.K., 2000. Effects on birds of an offshore wind park at Horns Rev: Environmental impact assessment. NERI report, MEE-NERI. <http://www.hornsrev.dk/miljoeforhold/miljoerapporter/Baggrundsrapport_18.pdf> (accessed 28.02. 12)

Orians, G.H., Pearson, N.E., 1979. On the theory of central place foraging, in: Horn, D.J., Mitchell, R.D., Stairs, G.R. (eds) (Eds.), Analysis of Ecological Ecosystems. Colombus: Ohio State University Press, pp. 155-177.

Oro, D., 1998. Larus audouinii Audouin's Gull, BWP Update (pp. 47-61). Oxford University Press.

Oro, D., Bosch, M., Ruiz, X., 1995. Effects of trawl moratorium on the breeding paramenters of the Yellow-legged Gull Larus cachinnans. Ibis 137, 547-549

Oro, D., Pérez-Rodríguez, A., Martínez-Vilalta, A., Bertolero, A., Vidal, F., 2009. Interference competition in a threatened seabird community: A paradox for a successful conservation. Biological Conservation 142, 1830-1835.

Oro, D., Ruiz, X., 1997. Exploitation of trawler discards by breeding seabirds in the north-western Mediterranean: differences between the Ebro Delta and the Balearic Islands areas. ICES Journal of Marine Science: Journal du Conseil 54, 695-707.

Paiva, V.H., Ramos, J. a., Martins, J., Almeida, A., Carvalho, A., 2007. Foraging habitat selection by Little Terns Sternula albifrons in an estuarine lagoon system of southern Portugal. Ibis 150, 18-31.

Palomera, I., 1992. Spawning of anchovy Engraulis encrasicolus in the Northwestern Mediterranean relative to hydrographic features in the region. Marine Ecology Progress Series 79, 215-223.

Pedrocchi, V., Oro, D., González-solís, J., Ruiz, X., Jover, L., 2002. Differences in diet between the two largest breeding colonies of Audouin's gulls: the effects of fishery activities. Scientia Marina 66, 313-320.

Perrow, M.R., Gilroy, J.J., Skeate, E.R., Tomlinson, M.L., 2011. Effects of the construction of Scroby Sands offshore wind farm on the prey base of Little tern Sternula albifrons at its most important UK colony. Marine Pollution Bulletin 62, 1661-70.

Pinheiro, J.., Bates, D.M., 2000. Mixed-Effects Models in S and S-PLUS, Front matter. Springer-Verlag, New York.

R Development Core Team, 2008. R: A Language and Environment for Statistical Computing. Vienna Austria R Foundation for Statistical Computing.

Sierra, J., Sánchez-Arcilla, A., GonzálezDelRio, J., Flos, J., Movellán, E., Mösso, C., Martínez, R., Rodilla, M., Falco, S., Romero, I., 2002. Spatial distribution of nutrients in the Ebro estuary and plume. Continental Shelf Research 22, 361-378.

Taylor, L.R., 1961. Aggregation, Variance and the Mean. Nature 189, 732-735.

Taylor, L.R., Woiwod, I.P., 1980. Temporal Stability as a Density-Dependent Species Characteristic. The Journal of Animal Ecology 49, 209-224.

Taylor, L.R., Woiwod, I.P., 1982. Comparative Synoptic Dynamics. I. Relationships between Inter- and Intra-specific spatial and temporal variance/mean population parameters. The Journal of Animal Ecology 51, 879-906.

Ward, P., Zahavi, A., 1973. The importance of certain assemblages of birds as "information-centres" for food-finding. Ibis 115, 517-534.

Wiese, F.K., Montevecchi, W.., Davoren, G.., Huettmann, F., Diamond, A.., Linke, J., 2001. Seabirds at risk around offshore oil platforms in the North-west Atlantic. Marine Pollution Bulletin 42, 1285-1290.

Wood, S., 2006. Generalized additive models: an introduction with R. Chapman and Hall/CRC, Boca Raton.

Östman, O., 2002. Distribution of bird cherry-oat aphids (Rhopalosiphum padi (L.)) in relation to landscape and farming practices. Agriculture, Ecosystems & Environment 93, 67-71.

RESUM

Conèixer les estratègies d'alimentació dels depredadors marins és essencial per comprendre els factors intrínsecs que controlen la seva distribució, abundància i la seva funció ecològica en l'ecosistema marí. En el següent capítol, es va investigar per primera vegada els moviments de cerca d'aliment i els patrons d'activitat de la gavina corsa Larus audouinii mitjançant l'ús de dades de localització per satèl•lit a partir de vuit adults reproductors a la colònia principal de l'espècie a tot el món (el Delta de l'Ebre, Mediterrània NO). Les gavines marcades s'alimentaven a la zona marina propera a la colònia de cria (62% dels llocs d'alimentació) i a l'àrea terrestre del Delta de l'Ebre (principalment els camps d'arròs, el 38% dels llocs d'alimentació). Els patrons d'activitat de cerca d'aliment va canviant significativament al llarg del dia; El seu mínim va del capvespre fins a la primera meitat de la nit (19-1 h, el 32% dels llocs actius) i és més alt durant la resta del dia (1-19 h; 75,5 ± 4,3% ubicacions d'actives). Aquests resultats confirmen la plasticitat alimentària d'aquesta au marina i, en base a la informació anterior sobre els hàbits alimentaris d'aquesta espècie, hipotetitzem sobre com els seus patrons d'activitat temporal i l'ús que fa de l'hàbitat podrien estar associats amb variacions en la disponibilitat de recursos alimentaris marins (per exemple, les migracions verticals diàries dels peixos pelàgics) i de l'explotació dels recursos terrestres (per exemple, crancs de riu americà Procambarus clarkii).

JOURNAL REFERENCE

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PDF available in the Appendix, page 122

Foraging movements of Audouin's gull *(Larus audouinii)* in the Ebro Delta, NW Mediterranean: A preliminary satellite-tracking study

4

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ABSTRACT

Knowing the foraging strategies of marine predators is essential to understand the intrinsic factors controlling their distribution, abundance and their ecological function within the marine ecosystem. Here, we investigated for the first time the foraging movements and activity patterns of Audouin's gull *Larus audouinii* by using satellite-tracking data from eight breeding adults in the main colony of the species worldwide (Ebro Delta, NW Mediterranean). Tagged gulls foraged in the marine area close to the breeding colony (62% of foraging locations) and in the terrestrial area of the Ebro Delta (mainly rice fields; 38% of foraging locations). The foraging activity patterns changed significantly throughout the day; lower from dusk through the first half of the night (19-1 h; 32% of active locations) and higher during the rest of the day (1-19 h; 75.5±4.3% of active locations). These results confirm the foraging plasticity of this seabird and, based on previous information about the dietary habits of this species, we hypothesize how its time-dependent activity patterns and habitat use could be associated with variations in the availability of marine food resources (e.g. diel vertical migrations of pelagic fish) and the exploitation of terrestrial resources (e.g. American crayfish *Procambarus clarkii*).

INTRODUCTION

An important issue in the feeding ecology of marine predators is the degree of plasticity of their foraging behaviour. In general, specialist predators are constrained to forage on a specific habitat and time of day determined by a specific prey availability (Futuyma and Moreno, 1988; Krebs and Davies, 1993; Julliard et al., 2006). Under changing conditions of prey availability, specialists are able to dapt their foraging strategy by extending foraging range or time spent foraging (e.g. Oro et al., 1997; Lewis et al., 2001; Schwemmer and Garthe, 2008). By contrast, generalist predators have the ability to exploit different trophic resources and, consequently, they present higher plasticity in their foraging strategies (Krebs and Davies, 1993; Boyd et al., 2006; Julliard et al., 2006). This opportunistic behaviour allows generalists to modify their foraging strategies (i.e. exploited habitat, range or temporal patterns) according, for instance, to the varying degree of competition for food. Indeed, the foraging plasticity of marine predators has allowed these organisms to benefit from anthropogenic food resources (e.g. fisheries discards, refuse dumps or introduced prey species; Tablado et al., 2010; Ramos et al., 2011; Wagner and Boersma, 2011).

Amongst marine predators, the Audouin's gull *Larus audouinii* is a good example of an opportunist species that exhibits clear plasticity in its diet habits. This Mediterranean endemic species exploits small pelagic fish (their main prey, see Oro 1998 and references therein), but also alternative anthropogenic resources such as demersal or benthonic fish from fisheries discards or invasive freshwater crabs from terrestrial habitat (Oro et al., 1996a; Oro and Ruiz, 1997; Oro et al., 1999; Navarro et al., 2010). This opportunistic behaviour is especially relevant in breeding populations located in areas where diverse trophic resources are highly available (e.g. Oro and Ruiz, 1997; Oro et al., 1999; Navarro et al., 2010), which is the case of the breeding population located in the Ebro Delta (Fig.1. NW Mediterranean). This colony holds around 12000-13000 breeding pairs of Audouin's gull, ca. 65% of the total world population (Oro et al., 2009). The marine ecosystem of the Ebro Delta is one of the most important fishing grounds in the Mediterranean Sea, resulting in one of the largest fishing fleets in this region, which generates a high quantity of fisheries discards (Coll et al., 2008). Moreover, freshwater resources such as the invasive American crayfish *Procambarus clarkii* in the rice fields of the Ebro Delta are abundant and easily available (Gutierrez-Yurrita et al., 1999), providing an alternative and proficient trophic resource for the species (Oro et al., 1996b; Longoni, 2010; Navarro et al., 2010).

Although the diet habits of the Audouin's gull are well known (e.g. Oro et al., 1997; Pedrocchi et al., 2002; Sanpera et al., 2007; Navarro et al. 2010), detailed information on the foraging movements is biased toward studies based on ship surveys (e.g. Abelló and Oro,1998; Arcos et al., 2001; Abelló et al., 2003), which are strongly biased by the influenceof fishery discards and underestimate the importance of land habitat utilization. The only previous telemetric study (radio-tracking) already pointed to the apparent importance of the terrestrial habitat for the breeding population of the Ebro Delta colony (Mañosa et al., 2004).

Here, we present preliminary results of the first satellite-tracking study of Audouin's gull during the breeding season in its largest breeding colony (Ebro Delta). This paper aims to quantify the foraging range of Audouin's gull, evaluate the habitat utilization of marine and terrestrial areas and

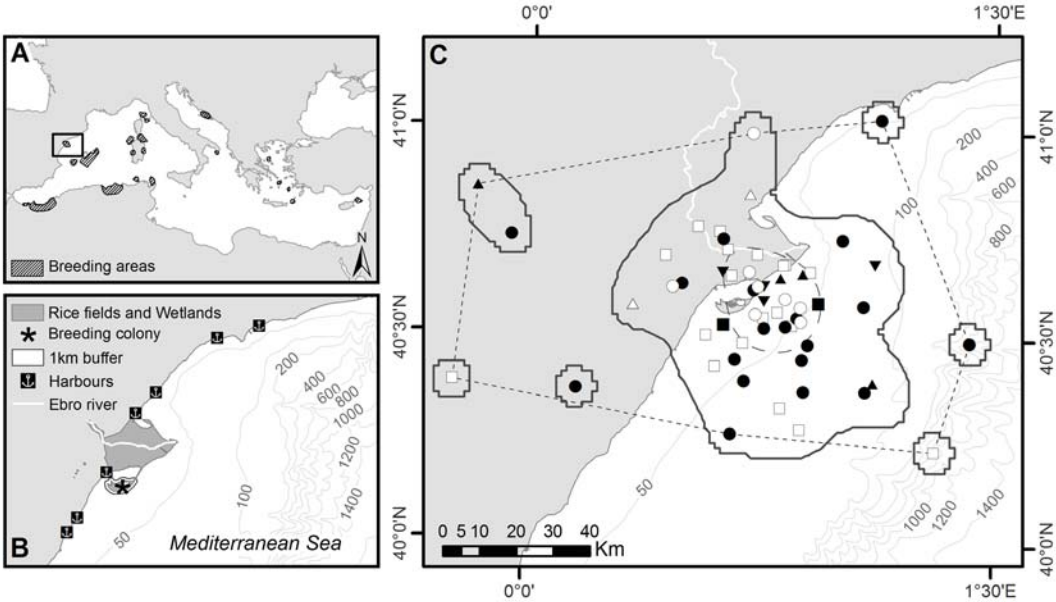


Fig.1 **(a)** *Breeding areas of the Mediterranean endemic Audouin's gull Larus audouinii and study area: Ebro Delta, NW Mediterranean. (BirdLife International, 2011)* **(b)** *Map of the Ebro Delta area indicating the Audouin's gull colony position with an asterisk and 1 km buffer area around la "Punta de la Banya" peninsula, the rice fields and wetlands shaded in dark gray and the location of the main harbours.* **(c)** *Foraging locations of 7 satellite-tracked Audouin's gulls during the breeding period of 2006. To better visualize the foraging locations' range the Minimum Convex polygon (short dashed line) is shown beside the 95% (solid line) and 50% (long dashed line) kernel polygons.*

identify the temporal patterns of the foraging activity of the species. Based on previous information about the dietary habits of this species, we also hypothesize how the observed foraging movements could be attributed to the exploitation of different trophic resources in the Ebro Delta marine and terrestrial ecosystems.

MATERIAL AND METHODS

Fieldwork procedures

The study was carried out at the natural reserve of Punta de la Banya in the Ebro Delta Natural Park, North Western Mediterranean Sea (Fig. 1, 40°33'N, 0°39'E). Punta de la Banya is a flat sandy peninsula of 2,514 ha, partially occupied by saltworks and connected to extensive rice field areas (20,000 ha) by a 5 km-long narrow sand bar. To examine the foraging activity, we satellite-tracked 8 breeding birds (4 males and 4 females) using battery powered "Platform Transmitter Terminals" (PTTs; North Star Science and Technology, LLC) during the chick-rearing period (May) of 2006 (Table 1). We captured all birds on the nest by using a drop trap (Mills and Ryder, 1979) during late incubation to reduce the risk of desertion. Once trapped, each individual was sexed, weighed, ringed and tagged with a PTT. The attached PTTs weighed 20 g and were programmed to be active in a 6 h on/5 h off duty cycle to get information on the foraging locations during one month. The PTT was fixed to the mid-dorsal feathers of the mantle using Tesa tape (Wilson et al., 1997). With this method the PTT falls off after one month without the necessity to recapture the instrumented bird. The entire transmitter equipment represented between 3 and 4% of the Audouin's gull's body mass, so the potential effects of an additional weight on the gull's movement were minimized (e.g. Phillips et al., 2003; Passos et al., 2010).

Table.1 *Summary information of PTTs performance*

PTT Id	Sex	Tracking days	First location	Last location	Total locations
58978	♂	2	15/05/2006	16/05/2006	6
58979	♂	1	19/05/2006	19/05/2006	2
58980	♂	10	18/05/2006	27/05/2006	31
58981	♂	2	15/05/2006	16/05/2006	4
58982	♀	7	15/05/2006	21/05/2006	6
58983	♀	0	-	-	-
58984	♀	10	15/05/2006	24/05/2006	32
58985	♀	3	18/05/2006	20/05/2006	8
Total		13	15/05/2006	27/05/2006	89

Satellite-tracking data and statistical analyses

Data on the position of each PTT were obtained from ARGOS system (CLS, Toulouse, France) and imported to ArcView 3.2 (ESRI) using the Argos Tool extension (Potapov and Dubinin, 2005). Each position was classified according to its estimated error: Type 0 (>1000 m), Type 1 (350-1000 m), Type 2 (150-350 m), Type 3 (0-150 m), and Types A and B (without an estimated error) (ARGOS, 2006). Initial data filtering involved calculating velocities between successive satellite locations, and rejecting those for which the velocity exceeded a threshold of 50 m·s-1, the maximum velocity described for this species (Rosén and Hedenström, 2001). By this procedure, up to 8 % of the locations were filtered; all of them from the low-quality accuracy class "B".

To gain insight into the foraging activity of the tagged Audouin's gulls we sorted the locations into three classes, according to their spatial position. PTT locations inside the "Punta de la Banya" peninsula or within the first kilometer around it were classified into the "colony locations" group. In contrast, the locations outside the colony and the first kilometer around it were "foraging locations" (we assumed that the birds were feeding to recover the body condition lost during the incubation bout). Finally, we calculated the 95% fixed-kernel estimates of the foraging area and the maximum foraging distance from the colony.

We employed logistic regression – a generalized linear model (GLM) – to test the foraging activity and habitat use. First, we tested a model with the proportion of foraging locations as the dependent variable, and we selected as the explanatory variable the "time of day" -categorized in 6-hour intervals (1-7 h; 7-13 h; 13-19 h; 19-1 h)- with the 7-13 h interval as the reference level. Then, we analyzed habitat use by testing the effect of the explanatory variable "time of day" on the dependent variable "terrestrial *vs.* marine proportion of foraging locations". The analyses were carried out using R software (R Development Core Team, 2008), calling the "glm" function with binomial error distribution and its default logit link function. A likelihood ratio test was used to compare the resulting model with the null model (without any variable) and to assess the significance of the explanatory variable "time of day".

RESULTS

We obtained a total of 89 filtered PTT locations spanning a period of 13 consecutive days. One of the eight PTTs failed to give any location probably due to a battery failure, and the performance of the remaining PTTs was heterogeneous (see Table 1). Due to sample size limitations individual variability was not included in the analysis, but the movements of one of the tracked individuals is shown in Figure 2 to illustrate the general pattern of the foraging movements.

The foraging area covered by the Audouin's gulls was 5400 km^2 (95% fixed-kernel density estimate), covering both the marine area of the Ebro Delta (ca. 3300 km^2) and the terrestrial area (ca. 2100 km^2) (Fig. 1c). The maximum foraging distance covered ranged from 20.5 to 81.7 km (mean ± sd = 51.5 ± 24.3 km) and was similar for both marine and terrestrial locations (T-Student test, T = 1.44, df = 56, p = 0.15).

The foraging activity changed significantly over the course of the day (Likelihood Ratio Test, χ^2 =13.79, df =3, p=0.003). Tagged gulls were more active at 7-13 h (78.1%), at 1-7 h (77.8% of the total locations in this period, p = 0.65), and 13-19h (70.6%, p = 0.56), all of them significantly different from the 19-1 h interval (31.8%, p= 0.001), i.e., the foraging activity diminished during the first half of the night (Fig. 3b). Moreover, we found that the proportion of foraging locations in marine *vs.* terrestrial habitats changed during the day. Although the time of day was not significant as a global explanatory variable, the model indicated a significant difference between the 13-19 h interval and the reference level 7-13 h (p= 0.04) (Fig. 3c). Between 13h and 19 h, Audouin's gulls foraged mainly in terrestrial (41%) rather than marine habitat (29%); during the rest of the day, they foraged mainly in marine rather than terrestrial habitat (1-7h: 50% marine, 28% terrestrial habitat; 7-13h: 59% marine, 19% terrestrial habitat; 19-1h: 23% marine, 9% terrestrial habitat)(Fig. 3a).

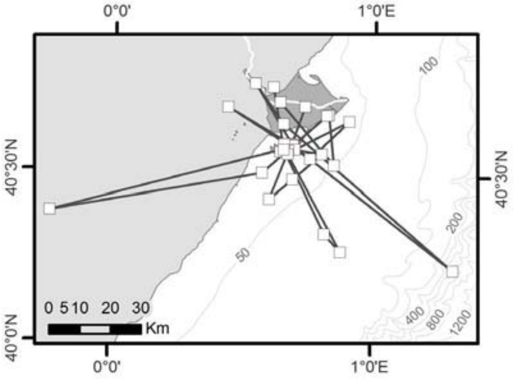


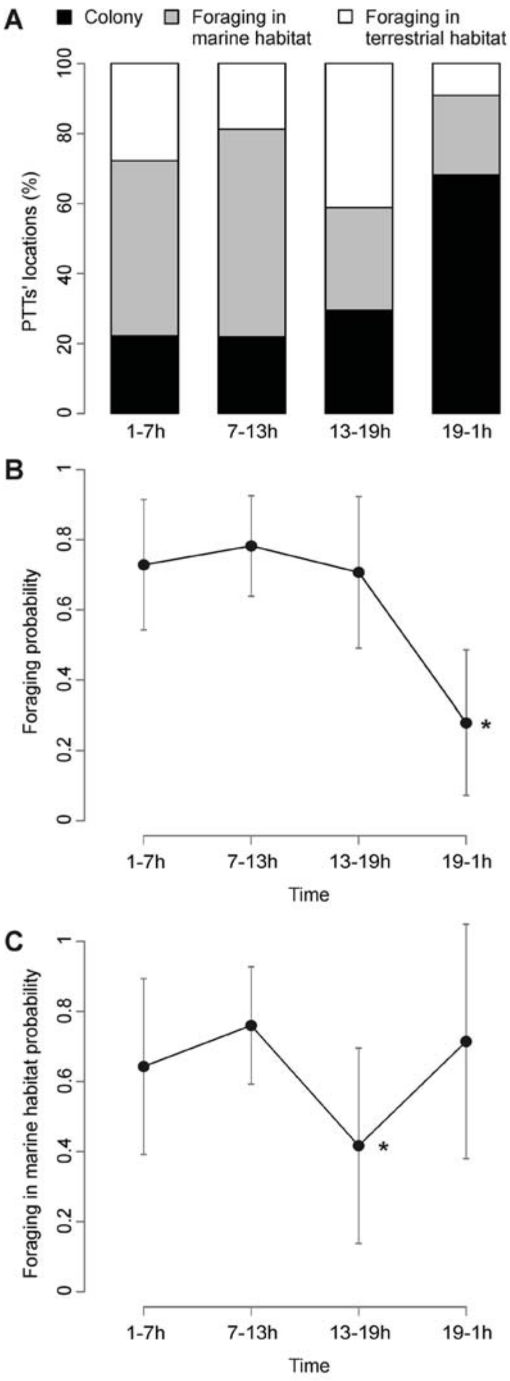
Fig.2 *Example of foraging trajectories for the individual "58980" (see Table 1 for more information)*

DISCUSSION

Satellite-tracked Audouin's gulls covered a foraging area that ranges 80 km, spanning both marine and terrestrial habitats. It has been widely described previously that breeding Audouin's gulls cover large ranges when foraging. There are records of individuals foraging at 70 to 150 km from the breeding colony during the breeding season (Baccetti et al., 2000; Mañosa et al., 2004), and data from vessel counts suggest that individuals forage during the day and night even further offshore (Abelló and Oro, 1998; Arcos and Oro, 1996). However, the species' terrestrial foraging movements had been scarcely described (Ruiz et al., 1996; Mañosa et al., 2004).

It is well documented that Audouin's gulls forage during the night in marine habitats preying on small pelagic fish and exploiting discards provided by nocturnal fisheries (e.g. Witt et al. 1981; Mañosa et al. 2004; Arcos et al., 2008). However, our results highlight that the species' nocturnal activity is not homogeneous throughout

Fig.3 (a) *Activity (foraging in marine or terrestrial habitat; or located in the colony) during a 24h cycle of 7 satellite-tracked Audouin's gulls during the breeding period in Ebro Delta colony.* **(b)**, **(c)** *Mean and 95% confidence interval, of the foraging probability and foraging in marine habitat probability respectively, according to the GLM models. * indicates a significant difference of the time block probability compared to the reference level 7-13 h.*



the night (see Fig.3). Satellite-tracked gulls were mainly located in the breeding colony during the hours before and after dusk (19-1 h). In the period after midnight to dawn (1-7 h) they increased their foraging activity, which then remained constant and high during the day. These results, coupled with the nocturnal arrival and departure times from the breeding colony described in Mañosa et al. (2004), confirm a peak of activity between midnight and dawn. Attendance to purse seiners during the night is considered a strategy that is only significant during trawling moratorium and winter periods (Arcos and Oro, 2002), neither of which were covered during our study; therefore, the individuals located at sea during the night were probably feeding on small pelagic fish. Accordingly, the nocturnal foraging habits of the Audouin's gull would still rely on the capture of small pelagic fish (Witt et al., 1981; Oro, 1998), a resource that might not be available throughout the night, but only in the hours before dawn due to the diel vertical migration of the shoals (Blaxter and Hunter, 1982; Oro, 1998).

With regard to diurnal activity, tagged birds showed a high foraging activity with an unexpected constant presence in terrestrial habitats (generally rice fields or wetlands) in addition to the expected presence in marine habitat (Oro, 1998). The fact that all tagged individuals could be found in both habitats suggests that the use of terrestrial habitat was not due to the casual behaviour of a single individual. This result supports previous studies that describe the use of the rice fields of the Ebro Delta by the Audouin's gull (Ruiz et al., 1996; Mañosa et al., 2004; Longoni, 2010), probably related to the exploitation of the exotic American crayfish (Navarro et al., 2010), which is very abundant in the rice fields of the Ebro Delta (Gutierrez-Yurrita et al., 1999). Although many studies have demonstrated that the Audouin's gull exploits trawler discards (Oro et al., 1997; Arcos, 2001; Cama, 2010), the foraging activity of our satellite-tracked individuals was higher inland than at sea in a period of time that includes the discarding peak of the trawling fleet (from 15 to 16 h; Cama, 2010). This result suggests that terrestrial foraging has become an alternative food source to trawling discards (Navarro et al., 2010), probably prompted by the interference competition for fisheries discards: namely, intraspecific competition (due to an increasing population density), and interspecific competition with the sympatric and dominant Yellow legged gull *Larus michahellis* (e.g. Arcos et al., 2001).

In conclusion, the present study shows that Audouin's gull foraged in both marine and terrestrial habitats and showed activity during both night and day. These results confirm the high foraging plasticity of Audouin's gull, a species once defined as a specialist nocturnal forager that has become an opportunist on fisheries discards and terrestrial resources. However, due the limited sample size we suggest the necessity of conducting more studies using biologging methodologies (such as PTTs or GPS) to confirm the observed patterns and to gain new insight into the foraging ecology of this endangered seabird.

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REFERENCES

Abelló, P., Oro D., 1998. Offshore distribution of seabirds in the northwestern Mediterranean in June 1995. Colonial Waterbirds 21, 422-426.

Abelló, P., Arcos, J., Gil Sola, L., 2003. Geographical patterns of seabird attendance to a research trawler along the Iberian Mediterranean coast. Scientia Marina 67, 69–75.

Arcos, J., 2001. Foraging ecology of seabirds at sea: significance of commercial fisheries in the NW Mediterranean. PhD Thesis, University of Barcelona.

Arcos, J., Oro, D., 1996. Changes in foraging range of Audouin's gulls *Larus audouinii* in relation to a trawler moratorium in the western Mediterranean. Colonial Waterbirds 1, 128-131.

Arcos, J., Oro, D., Sol, D., 2001. Competition between the yellow-legged gull *Larus cachinnans* and Audouin's gull *Larus audouinii* associated with commercial fishing vessels: the influence of season and fishing fleet. Marine Biology 139, 807-816.

Arcos, J., Oro, D., 2002. Significance of nocturnal purse seine fisheries for seabirds: a case study off the Ebro Delta (NW Mediterranean). Marine Biology 141, 277–286.

Arcos, J., Louzao, M., Oro, D., 2008. Fishery ecosystem impacts and management in the mediterranean: seabirds point of view. American Fisheries Society Symposium 49, 1471-1479.

ARGOS, 2006. Argos User's Manual. ARGOS CLS, Toulouse, France. http://www.argos- system.org/manual/link. Last accessed 24 November 2011.

Baccetti, N., Dall'Antonia, L., Magnani, A., Serra, L., 2000. Foraging routes of Audouin's Gulls *Larus audouinii* from two Sardinian colonies. In: Sultana, J., Yésou, P. (Eds.), Monitoring and conservation of birds, mammals and sea turtles of the Mediterranean and Black Seas. Environment Protection Department, Malta, pp. 150-158.

BirdLife International, 2011. Distribution maps of Birds of the World. BridLife International, Cambridge, UK.

Blaxter, J.H.S., Hunter, J.R., 1982. The biology of clupeoid fishes. Advances in Marine Biology 20, 1-223.

Boyd, I.L., Wanless, S., Camphuysen, C.J. (Eds.), 2006. Top predators in marine ecosystems: their role in monitoring and management. Cambridge University Press, Cambridge, 378 pp.

Cama, A., 2010. Seabird distribution patterns and ecological factors driving larid presence at the Ebro delta shelf (NW Mediterranean). PhD Thesis, University of Barcelona.

Coll, M., Palomera, I., Tudela, S., Dowd, M., 2008. Food-web dynamics in the South Catalan Sea ecosystem (NW Mediterranean) for 1978–2003. Ecological Modelling 217, 95-116.

Futuyma, D.J., Moreno, G., 1988. The evolution of ecological specialization. Annual Review of Ecology and Systematics 19, 207–233.

Garthe, S., Camphuysen, C.J., Furness, R.W., 1996. Amounts discarded by commercial fisheries and their significance as food for seabirds in the North Sea. Marine Ecology Progress Series 136, 1-11.

Gutierrez-Yurrita, P., Martínez, J., Ilhéu, M., Bravo-Utrera, M., Bernardo, J., Montes, C., 1999. The status of crayfish populations in Spain and Portugal. In: Gerardi, F., Holdich, D. (Eds.), Crayfish in Europe as alien species: How to make the best of a bad situation?. Crustacean Issues, Balkema, Rotterdam, pp. 161–192.

Julliard, R., Clavel, J., Devictor, V., Jiguet, F., Couvet, D., 2006. Spatial segregation of specialists and generalists in bird communities. Ecology letters 9, 1237-44.

Krebs, J.R., Davies, N.B., 1993. An introduction to behavioural ecology. Wiley-Blackwell publishing, Oxford, 420 pp.

Lewis, S., Sherratt, T.N., Hamer, K.C., Wanless, S., 2001. Evidence of intra-specific competition for food in a pelagic seabird. Nature 412, 816-819.

Longoni, V., 2010. Rice Fields and Waterbirds in the Mediterranean Region and the Middle East. Waterbirds 33, 83-96.

Mañosa, S., Oro, D., Ruiz, X., 2004. Activity patterns and foraging behaviour of Audouin's gulls in the Ebro Delta, NW Mediterranean. Scientia Marina 68, 605-614.

Mills, J.A., Ryder, J.P., 1979. Trap for Capturing Shore and Seabirds. Bird-Banding, 50, 121-123.

Navarro, J., Oro, D., Bertolero, A., Genovart, M., Delgado, A., Forero, M.G., 2010. Age and sexual differences in the exploitation of two anthropogenic food resources for an opportunistic seabird. Marine Biology 157, 2453-2459.

Orians, G.H., Pearson, N.E., 1979. On the theory of central place foraging. In: Horn, D.J., Stairs, G.R., Mitchell, R.D. (Eds.), Analysis of Ecological Ecosystems. Ohio State University Press, Colombus, pp. 155-177.

Oro, D., 1998. *Larus audouinii* Audouin's Gull. BWP Update. Oxford University Press, Oxford, pp. 47-61

Oro, D., Jover, L., Ruiz, X., 1996a. Influence of trawling activity on the breeding ecology of a threatened seabird, Audouin's gull *Larus audouinii*. Marine Ecology Progress Series 139, 19-29.

Oro, D., Genovart, M., Ruiz, X., Jiménez, J., García-Gans, J., 1996b. Differences in diet, population increase and breeding performance between two colonies of Audouin's gulls *Larus audouinii* during breeding seasons affected by a trawling moratorium. Journal of Avian Biology 27, 245-251.

Oro, D., Ruiz, X., 1997. Exploitation of trawler discards by breeding seabirds in the north-western Mediterranean: differences between the Ebro Delta and the Balearic Islands areas. ICES Journal of Marine Science 54, 695-707.

Oro, D., Ruiz, X., Jover, L., Pedrocchi, V., González-Solís, J., 1997. Diet and adult time budgets of Audouin's gull *Larus audouinii* in response to changes in commercial fisheries. Ibis 139, 631-637.

Oro, D., Pradel, R., Lebreton, J.D., 1999. Food availability and nest predation influence life history traits in Audouin's gull, *Larus audouinii*. Oecologia 118, 438-445.

Oro, D., Cam, E., Pradel, R., Martínez-Abraín, A., 2004. Influence of food availability on demography and local population dynamics in a long-lived seabird. Proceedings of the Royal Society of London 271, 387-396.

Oro, D., Pérez-Rodríguez, A., Martínez-Vilalta, A., Bertolero, A., Vidal, F., 2009. Interference competition in a threatened seabird community: A paradox for a successful conservation. Biological Conservation 142, 1830-1835.

Passos, C., Navarro, J., Giudici, A., González-Solís, J., 2010. Effects of Extra Mass on the Pelagic Behaviour of a Seabird. The Auk 127, 100-107.

Pedrocchi, V., Oro, D., González-Solís, J., Ruiz, X., Jover, L., 2002. Differences in diet between the two largest breeding colonies of Audouin's gulls: the effects of fishery activities. Scientia Marina 66, 313-320.

Phillips, R., Xavier, J.C., Croxall, J.P., Burger, A.E., 2003. Effects of satellite transmitters on albatrosses and petrels. The Auk 120, 1082-1090.

Potapov, E., Dubinin, M., 2005. Argos-tools manual. The Falcon Research Institute, National Avian Research Center, Carmarthen, UK.

Ramos, R., Ramírez, F., Carrasco, J.P., Jover, L., 2011. Insights into the spatiotemporal component of feeding ecology: an isotopic approach for conservation management sciences. Diversity and Distributions 17, 1-12.

R Development Core Team, 2008. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org/.

Rosén, M., Hedenström, A., 2001. Testing predictions from flight mechanical theory : a case study of Cory's shearwater and Audouin's gull. Acta ethologica 3, 135-140.

Ruiz, X., Oro, D., Martínez-Vilalta, A., Jover, L., 1996. Feeding ecology of Audouin's gull *Larus audouinii* in the Ebro Delta. Colonial Waterbirds 19, 68-74.

Sanpera, C., Ruiz, X., Moreno, R., Jover, L., Waldron, S., 2007. Mercury and stable isotopes in feathers of Audouin's gulls as indicators of feeding habits and migratory connectivity. The Condor 109, 268-275.

Schwemmer, P., Garthe, S., 2008. Regular habitat switch as an important feeding strategy of an opportunistic seabird species at the interface between land and sea. Estuarine, Coastal and Shelf Science. 77, 12-22.

Tablado, Z., Tella, J.L., Sánchez-Zapata, J.A., Hiraldo, F., 2010. The paradox of the long-term positive effects of a North American Crayfish on a European community of predators. Conservation biology 24, 1230-1238.

Wagner, E.L., Boersma, PD., 2011. Effects of fisheries on seabird community ecology. Reviews in Fisheries Science 19, 157-163.

Wilson, R.P., Pütz, K., Peters, G., Culik, B., Scolaro, J.A., Charrassin, J.B., Ropert-Coudert, Y., 1997. Long-term attachment of transmitting and recording devices to penguins and other seabirds. Wildlife Society Bulletin 25, 101-106.

Witt, H.H., Crespo, J., de Juana, E., Varela, J.M., 1981. Comparative feeding ecology of Audouin's Gull *Larus audouinii* and the Herring Gull *L. argentatus* in the Mediterranean. Ibis 123, 519-526.

Discussion & Conclusions

DISCUSSION

“Gaining insight: What I learned from the experience”

SENSITIVITY INDEX

The work of Garthe and Hüppop (2004) proposed the quantitative assessment of the vulnerability of seabird community to wind farms. The index framework calculated this vulnerability through the Species Sensitivity Index (SSI) that focuses on the species vulnerability (at individual and population levels) and the Wind Farm Sensitivity Index (WSI) that applies the SSI to estimate the community vulnerability. However, as it has been shown in Chapter 1, the mathematical formulation of the original index contains hidden assumptions at both species and community levels that might lead to incorrect estimates of vulnerability and a biased identification of key areas.

The first assumption was that all risk factors associated to a given type of risk had equal importance and had an additive relationship. Nevertheless, there is a conceptual difference between the factors included in a particular type of risk. Taking collision risk as an example, we find two types of risk factors: those directly associated to the risk itself (*e.g.* percentage of time spent at high altitude when flying) and factors that are not important in themselves, but as an aggravation factor that can increase a risk if it already exists (*e.g.* flight manoeuvrability). In other words, if a seabird species never flies within a height with risk of collision, it is irrelevant the manoeuvrability of the species since it will never have to avoid a turbine. This means that factors have a hierarchical structure between primary risk factors and aggravation factors that cannot be dealt with an additive formulation; therefore an alternative power function is suggested to estimate collision and disturbance risk.

The second assumption was that each type of risk (collision, disturbance and population sensitivity) was equally important and had a multiplicative relationship. It is difficult to measure the relative importance of collision risk over disturbance risk, which justifies considering them as equal by default despite collision mortality is often considered to be the most important hazard (Fox et al., 2006; Christel, Certain, et al., 2012). Having a multiplicative relationship between collision and disturbance risk is conflictive because these two types of risk do not depend on each other. They are two independent aspects of the potential impact of an OWF on seabirds and thus, collision and disturbance impact have different ecological consequences. By linking them with a multiplicative relationship might result in the underestimation of a risk effect, only because the other risk is very low. In fact, this is the case in Garthe and Hüppop's calculation of the SSI which is more correlated to the disturbance risk than to the collision risk. If the two types of risks have to be combined in a single map I would recommend using an additive relationship. But I don't recommend such combination but rather the individual examination of both collision risk maps and disturbance risk maps to reach an informed decision for management.

The final step of the index is to integrate the species vulnerability into a measure of vulnerability of a whole community. To do so, the third assumption was that the contribution to the community vulnerability of a given seabird species is proportional to the log abundance of that species in a particular location. The rationale for this approach was to prevent the installation of wind farms in areas with high aggregations of seabirds. However, abundant species are usually those with lower SSI values while rare species are those with higher SSI values. For a given location, SSI and abundance affect the final WSI value in opposite directions which hinders the interpretation of the variations of the index. Moreover, the use of log abundance instead of plain abundance assumes that a single individual has more weight, in proportion, than an individual located in a flock of one hundred seabirds, which has neither ecological nor environmental support. To solve these inconsistencies, a major change in the formulation is suggested based on the work of Leinster and Cobbold (2012) that presents a diversity measure based not only on species abundance but also on the similarity of these species.

After incorporating the changes in the index formulation and applying the index to a large study area like the coasts of the Iberian Peninsula, there are some conclusions and recommendations that can be drawn from the experience.

This type of index is eminently comparative therefore it is better to apply it in the broader scale possible. Iberian Peninsula is a geographical unit and regardless the borders it is interesting to see how it is in the conjunct. Based on boat surveys means maximum probabilities fo detecting all species which ensures fiability in the index.

Placement decision on the trade-off between impact and benefits is not a scientific but managers decision. Although Garthe et al. set a criterion for defining concern and major concern we think that it is better to rank the output accordingly to their WSI avoiding delimitations of a subjective value. That moreover can be different according to the average WSI in the area.

By applying the index at different scales we find consistent patterns that remain constant through the scales although loosing detail with larger scales. However it shows that different scales can be used to different management purposes. The utilization of a large scale WSI grid, with grid cells between 1° and 0.5°, seems more appropriate for the definition of optima development areas, while small scale WSI grids, e.g. 0.25°, are better for the demarcation of areas of high vulnerability or areas of high concern if there was a project to install an OWF within them. Highlight the difference between optimal development areas and priority areas for conservation regarding seabirds and OWF. Small scale application of the WSI seems optimal for hazard location and the definition of high risk areas that must be specifically protected to be preserved from the construction of any OWF and the indirect influence of the cumulative impact of OWF.

Although this method is an integrative tool and the example of the study area shows it's utility, any technique based on boat or aerial surveys has some methodological limitations that have been described in the methodological approach section. Because of this, it is "recommendable" to complement these indexes with new methods like satellite tracking to complement the information with the offshore distribution of flagship species. In the case presented, there is the example of the Cape of Naos (Spanish Mediterranean coast in front of Ibiza) which is an area of extensive use of Puffinus mauretanicus but can't be detected by the boat surveys because their activity peak is out of the temporal scope of the boats surveys.

AGGREGATIVE PATTERNS

Abundance maps define areas with high numbers of birds, which is relevant information in the assessment of offshore wind farms locations, but they do not provide information on the dynamic and complex spatial patterns of seabirds at sea. While density maps focus on the detection of high concentrations of seabirds as potential risk areas, the application of Taylor's power law allows the explicit distinction between transitional and foraging areas over time. Taylor's power law is widely accepted as an aggregation index in time and space, which is corroborated in Chapter 3 by its correlation with mean flock size. Areas with weak aggregative patterns can be considered transitional or flight path areas while high aggregative patterns are mostly feeding areas determined by the punctual availability of a feeding opportunity.

By linking the aggregative patterns with a particular behaviour we can better predict and classify the risk of wind farm establishment on a seabird population or community. In transitional areas, the main risk will be direct collision and mortality (Desholm and Kahlert, 2005; Hüppop et al., 2006). In foraging areas, the presence of turbines would result in habitat loss for species with a strong avoidance response or an increased collision risk for the species that experience a low 'barrier effect' (Masden, Haydon, et al., 2010; Perrow et al., 2011). Therefore, after quantifying the aggregative pattern in a given area, the potential risk can be evaluated and used to rank regions within this area according to different levels of concern. Areas with both Collision and Habitat loss risk would have the higher concern level (L3); areas with collision risk would have the next concern level (L2) followed by areas with habitat loss risk (L1). This classification can be used later to choose the optimal location of an OWF (by choosing areas with minimum concern) or to define a required monitoring protocol for an accepted OWF location according to the concern level of this location. Moreover, there is a temporal evolution of the aggregative patterns and it is correlated to the life cycle of the species. Whether we use seasonal scenarios (*e.g.* breeding, post-breeding, migration, wintering) or a set of critical months, temporal scenarios are easier to communicate because they summarize key information that can highlight the potential impact of an OWF in a sensitive moment in a seabird life cycle or can be used to recommend mitigation measures during critical months.

The application of this tool and the interpretation of its results are particularly true for the assessment of areas with large breeding populations. However, it would be advisable to apply this method in other scenarios to investigate any possible differences in the interpretation of the potential risks associated to the observed aggregative patterns. For instance, areas with a different composition of species like plunge-divers (*e.g.* gannets) or surface-divers (*e.g.* sea ducks) might show different spatial and temporal patterns. It would also be desirable to study the outputs of the tool in a migratory corridor. Finally, although the method results are consistent with the ornithological observations and behavioural data in the Ebro Delta, I would encourage the application of telemetry data to test with an independent data set the behavioural interpretation of the Taylor's power law.

Nevertheless, the study presented in Chapter 3 demonstrates that considering the information on the second order properties of species' distributions (*i.e.* the social aggregation) provides further information for the assessment of potential impacts of offshore wind farms than solely focusing on the first order properties (*i.e.* density).

INDIVIDUAL TRACKING

Seabird locations and seabird behaviour are distinct, and the latter is an important component that can be extracted from individual tracking data types (Tremblay et al., 2009). One of the aims of Chapter 4 was to perform State-Space Models (SSM; Jonsen et al., 2003) on satellite tracking data as a final alternative on spatio-temporal assessment tools. However, the performance of the Platform Transmitter Terminals (PTTs) was poor and heterogeneous among devices, probably due to battery problems. The final sample size was very low and the time span between locations too long to perform State-Space Models or to test the conclusions of Chapter 3.

Despite the technical problems and limited sample size, some general conclusions can be drawn from the results of the spatial and temporal analysis of the movements of the Audouin's Gull *Larus audouinii*. When foraging, seabirds have to overcome the variability on the distribution, abundance, mobility and predictability of their food sources (Bell, 1991). To do so, seabird species show a certain degree of plasticity on their at-sea behaviour. Among many possible behavioural responses (*e.g.* trip duration, travelling distances, diving patterns) Chapter 4 draws attention to the plasticity on the temporal activity patterns and the habitat use and this plasticity has to be addressed in the assessment of any offshore wind farm.

Many studies have shown that the activity pattern of seabirds is not constant throughout the day (*e.g.* Garthe et al., 2003; Cama, 2010; Cama et al., 2012; Christel, Navarro, et al., 2012). Therefore, seabird surveys, which must be performed with daylight and usually following a constant schedule, are not always sufficient to capture the variability of seabird circadian cycles. Some seabirds, for instance, relay on small pelagic fish, but this resource might only be available in the hours before dawn when the shoals perform their vertical migration (Blaxter and Hunter, 1982). In this case, an area with recurrent aggregations of seabirds foraging on these shoals wouldn't be detected by surveys which usually start after dawn.

The foraging plasticity of seabirds, regarding their habitat use and prey selection, is sometimes underestimated. In conditions of reduced prey availability, specialised seabird species usually modify their feeding strategy by extending their foraging area, the time spent at sea or reducing the time between trips (*e.g.* Lewis et al., 2001; Schwemmer and Garthe, 2008) and generalist species may change their foraging habitats or shift their diet (e.*g.* González-Solís et al., 1997; Schwemmer and Garthe, 2008; Navarro et al., 2010).

CONCLUSIONS

“It always takes awfully long time to understand unbelievably simple things” Joe Chung

1 Garthe and Hüppop method (2004) is general, simple and widely applicable; hence instead of developing a new index it is better to refine the existing one. The mathematical formulation of the original index contains hidden assumptions at both species and community levels that might lead to incorrect estimates of vulnerability and a biased identification of key areas.

2 The refined framework to amend the problematic assumptions includes de distinction between Direct and Aggravation factors within a Risk type; the independent identification of collision risk areas and disturbance risk areas; and the incorporation of recent developments in functional diversity to produce a vulnerability map based on local relative frequencies of species.

3 The refined approach proposed in this paper has a much larger potential of application than the restricted scope of offshore wind farm impact assessment. It can in fact be applied to estimate the vulnerability of any kind of community to any kind of impact, provided that a measure of the species-specific vulnerability to that impact is defined and community distribution data has been collected.

4 The application of the refined index at different grid scales can be used to different management purposes. The utilization of a large scale WSI grid, is more appropriate for the definition of optima development areas, while small scale WSI grids are better for the demarcation of areas of high vulnerability if there was a project to install an OWF within them.

5 This type of index is eminently comparative therefore it is better to apply it in the broader scale possible Small scales for hazard location and the definition of risk areas. Placement decision on the tradeoff between potential impacts and benefits and it is not scientists but managers' decision. Although Garthe and Hüppop set a criterion for defining concern and major concern areas we think that it is better to rank the output accordingly to their WSI avoiding the definition of a subjective threshold value.

6 Abundance maps define areas with high numbers of birds, which is relevant information in the assessment of offshore wind farms locations, but they do not provide information on the dynamic and complex spatial patterns of seabirds at sea. Taylor's power law slope can measure seabirds' aggregative pattern in time and space. That can be used to highlight recurrent transitional and feeding areas.

7 In transitional areas, the main risk will be direct collision and mortality. In foraging areas, the presence of turbines would result in habitat loss for species with a strong avoidance response or an increased collision risk for the species that experience a low 'barrier effect'. This information can be used to inform on concern levels, optimal EIA design and monitoring in the assessment of local offshore wind energy projects.

8 When foraging, seabirds have to overcome the variability on the distribution, abundance, mobility and predictability of their food sources. To do so, seabird species show a certain degree of plasticity on their at-sea behavior, particularly plasticity on the temporal activity patterns and the habitat use.

9 Seabird surveys are constrained in space by the arrangement of the survey transects, and constrained in time by the moment of the day at which each transect is surveyed and the necessity of daylight for the counts. Individual tracking of seabirds, instead, is not restricted in either space or time. By using telemetry techniques besides surveys the variations on the species behavior in a 24h cycle can be assessed and included in the assessment of the potential impacts of the presence of offshore wind farms.

10 The foraging plasticity in seabird species is usually underestimated, individual-based studies may show individual differences in habitat use, the exploitation of alternative food sources out of the concern areas and the potential capability of species to switch their foraging grounds, should the individuals find a barrier in their preferential habitats.

11 Given this potential variability at individual level, any assessment study based on aerial or boat surveys should be complemented with telemetry data, selecting for this purpose high concern species or flagship species of the area to overcome the limitations of any systematic survey method.

12 To integrate the presented tools in the decision making process for offshore wind energy development we recommend the use of large scale visualizations of the Wind farm sensitivity index to define optimal development areas and the use of small scale visualizations of the Wind farm sensitivity index to avoid high concern areas. Once the development regions have been selected Taylor's power law analysis of the aggregative patterns should be implemented to map de potential impacts on the region in order to inform on the optimal location that minimizes de concern and state the compulsory monitoring programs of a location before selecting it. Finally it is advisable to identify flagship or high concern species in the area and perform telemetric studies to complement the distributional information in order to overcome the methodological limitations of the surveying techniques.

REFERENCES

> *"Science is a sophisticated block building game of knowledge"*

Allen, T., Hoekstra, T., 1991. Role of Heterogeneity in Scaling of Ecological Systems Under Analysis.

Anderson, D., Huyvaert, K., Wood, D., 2003. At-sea distribution of waved albatrosses and the Galapagos Marine Reserve. Biological Conservation 110, 367–373.

Arcos, J., 2001. Foraging ecology of seabirds at sea: Significance of commercial fisheries in the NW Mediterranean. Thesis.

Arcos, J., Louzao, M., Oro, D., 2008. Fisheries ecosystem impacts and management in the Mediterranean: seabirds point of view. American Fisheries Society Symposium 49, 1471–1479.

Arcos, J.M., Bécares, J., Villero, D., Brotons, L., Rodríguez, B., Ruiz, A., 2012. Assessing the location and stability of foraging hotspots for pelagic seabirds: An approach to identify marine Important Bird Areas (IBAs) in Spain. Biological Conservation.

Bailey, H., Senior, B., Simmons, D., Rusin, J., Picken, G., Thompson, P.M., 2010. Assessing underwater noise levels during pile-driving at an offshore windfarm and its potential effects on marine mammals. Marine Pollution Bulletin 60, 888–897.

Barry, S.C., Welsh, a. H., 2002. Generalized additive modelling and zero inflated count data. Ecological Modelling 157, 179–188.

Bell, W.J., 1991. Searching behaviour. Chapman & Hall, London.

Bellier, E., Certain, G., Planque, B., Monestiez, B., Bretagnolle, V., 2010. Modelling habitat selection at multiple scales with multivariate geostatistics: an application to seabirds in open sea. Oikos 119, 988–999.

Bigas, D., 2012. Llista Patró dels ocells del delta de l'Ebre. Categories A, B i C. Categories D i E. Parc Natural del Delta de l'Ebre, Departament d'Agricultura, Ramaderia, Pesca, Alimentación i Medi Natural, Generalitat de Catalunya.

BirdLife International, 2003. Windfarms and Birds: An analysis of the effects of windfarms on birds, and guidance environmental assessment criteria and site selection issues. Report written by BirdLife International on behalf of the Bern Covention.

Blaxter, J.H.S., Hunter, J.R., 1982. The biology of clupeoid fishes. Advances in Marine Biology 20, 1–223.

Boyd, I.L., Wanless, S., Camphuysen, C.J., 2006. Top predators in marine ecosystems: their role in monitoring and management. Cambridge University Press.

Bretagnolle, V., Certain, G., Houte, S., Métais, M., 2004. Distribution maps and minimum abundance estimates for wintering auks in the Bay of Biscay , based on aerial surveys. Aquatic Living Resources 353–360.

Broek, J.V. den, 1995. A score test for zero inflation in a Poisson distribution. Biometrics 51, 738–743.

Brookes, K., 2009. Assessment of methods used to investigate the impact of offshore wind farms on seabirds.

Burkhard, B., Opitz, S., Lenhart, H., Ahrendt, K., Garthe, S., Mendel, B., Windhorst, W., 2011. Ecosystem based modeling and indication of ecological integrity in the German North Sea—Case study offshore wind parks. Ecological Indicators 11, 168–174.

Cabal, J., González-Nuevo, G., Nogueira, E., 2008. Mesozooplankton species distribution in the NW and N Iberian shelf during spring 2004: Relationship with frontal structures. Journal of Marine Systems 72, 282–297.

Cairns, D., 1987. Seabirds as indicators of marine food supplies. Biological Oceanography 5, 261–271.

Cama, A., 2010. Seabird distribution patterns and ecological factors driving larid presence at the Ebro delta shelf (NW Mediterranean). Thesis.

Cama, A., Abellana, R., Christel, I., Ferrer, X., Vieites, D.R., 2012. Living on predictability: modeling the density distribution of efficient foraging seabirds. Ecography 35, 912–921.

Camphuysen, C.J., Fox, A.D., Leopold, M.F., Petersen, I.K., 2004. Towards standardised seabirds at sea census techniques in connection with environmental impact assessments for offshore wind farms in the U.K., Cowrie Report. London.

Certain, G., Bellier, E., Planque, B., Bretagnolle, V., 2007. Characterising the temporal variability of the spatial distribution of animals: an application to seabirds at sea. Ecography 30, 695–708.

Certain, G., Bretagnolle, V., 2008. Monitoring seabirds population in marine ecosystem: The use of strip-transect aerial surveys. Remote Sensing of Environment 112, 3314–3322.

Certain, G., Ridoux, V., van Canneyt, O., Bretagnolle, V., 2008. Delphinid spatial distribution and abundance estimates over the shelf of the Bay of Biscay. ICES Journal of Marine Science 65, 656–666.

Chamberlain, D., Rehfisch, M., Fox, A.D., Desholm, M., Anthony, S., 2006. The effect of avoidance rates on bird mortality predictions made by wind turbine collision risk models. Ibis.

Christel, I., Certain, G., Cama, A., Vieites, D.R., Ferrer, X., 2012. Seabird aggregative response: a new tool for offshore wind energy risk assessment (In press). Marine Pollution Bulletin.

Christel, I., Navarro, J., del Castillo, M., Cama, A., Ferrer, X., 2012. Foraging movements of Audouin's gull (Larus audouinii) in the Ebro Delta, NW Mediterranean: A preliminary satellite-tracking study. Estuarine, Coastal and Shelf Science 96, 257–261.

Christensen, T.K., Hounisen, J.P., Clausager, I., Petersen, I.K., 2004. Visual and radar observations of birds in relation to collision risk at the Horns Rev Offshore Wind Farm.

Desholm, M., 2006. Wind farm related mortality among avian migrants.

Desholm, M., 2009. Avian sensitivity to mortality: prioritising migratory bird species for assessment at proposed wind farms. Journal of environmental management 90, 2672–9.

Desholm, M., Fox, A.D., Beasley, P.D.L., Kahlert, J., 2006. Remote techniques for counting and estimating the number of bird-wind turbine collisions at sea: a review. Ibis 148, 76–89.

Desholm, M., Kahlert, J., 2005. Avian collision risk at an offshore wind farm. Biology Letters 1, 296–298.

Drewitt, A., Langston, R.W., 2006. Assessing the impacts of wind farms on birds. Ibis 148, 29–42.

EC, 2011a. A Roadmap for moving to a competitive low carbon economy in 2050. COM (2011) 112 of 8 March 2011. European Commission.

EC, 2011b. Energy roadmap 2050. COM (2011) 885 of 15 December 2011. European Commission.

EWEA, 2011. European offshore wind projects map. 3rd Edition. November 2011. EWEA.

EWEA, 2012. The European offshore wind industry key 2011 trends and statistics. European Wind Energy Association.

Elliott, M., 2002. The role of the DPSIR approach and conceptual models in marine environmental management: an example for offshore wind power. Marine Pollution Bulletin 44, iii–vii.

Eurostat, 2009. Panorama of energy. Energy statistics to support EU policies and solutions (2009 Edition).

Fauchald, P., Erikstad, K.E., Skarsfjord, H., 2000. Scale-Dependent Predator-Prey Interactions: The Hierarchical Spatial Distribution of Seabirds and Prey. Ecology 81, 773.

Fauchald, P., Erikstad, K.E., Systad, G.H., 2002. Seabirds and marine oil incidents: is it possible to predict the spatial distribution of pelagic seabirds? Journal of Applied Ecology 39, 349–360.

Fox, A.D., Desholm, M., Kahlert, J., Christensen, T.K., Petersen, I.K., 2006. Information needs to support environmental impact assessment of the effects of European marine offshore wind farms on birds. Ibis 148, 129–144.

Fox, A.D., Petersen, I.K., 2006. Assessing the degree of habitat loss to marine birds from the development of offshore wind farms, in: Boere, G.C., Galbaraith, C.A., Stroud, D.A. (Eds.), Waterbirds Around the World. The Stationery Office, Edimburg, UK, pp. 801–804.

Garthe, S., Benvenuti, S., Montevecchi, W.A., 2003. Temporal patterns of foraging activities of northern gannets, Morus bassanus, in the northwest Atlantic Ocean. Canadian journal of Zoology 81, 453–461.

Garthe, S., Markones, N., Mendel, B., Sonntag, N., Krause, J.C., 2011. Protected areas for seabirds in German offshore waters: Designation, retrospective consideration and current perspectives. Biological Conservation.

Gerkema, T., Lam, F. a., Maas, L.R.M., 2004. Internal tides in the Bay of Biscay: conversion rates and seasonal effects. Deep-Sea Research II 51, 2995–3008.

Gill, A.B., 2005. Offshore renewable energy: ecological implications of generating electricity in the coastal zone. Journal of Applied Ecology 42, 605–615.

González-Solís, J., Oro, D., Jover, L., Ruiz, X., Pedrocchi, V., 1997. Trophic niche width and overlap of two sympatric gulls in the southwestern mediterranean. Oecologia 112, 75–80.

González-Solís, J., Shaffer, S., 2009. Spatiotemporal dynamics of seabirds in the marine environment. Marine Ecology Progress Series 391, 117–120.

Henderson, A.R., Morgan, C., Smith, B., Sorensen, H.C., Barthelmie, R.J., Boesmans, B., 2003. Offshore Wind Energy in Europe? A Review of the State-of-the-Art. Wind Energy 6, 35–52.

Henkel, L.A., Ford, R.G., Tyler, W.B., Davis, J.N., 2007. Comparison of aerial and boat-based survey methods for Marbled murrelets Brachyramphus marmoratus and other marine birds. Marine Ornithology 35, 145–151.

Hunt, G.L., Schneider, D.C., 1987. Scale-dependent processes in the physical and biological environment of marine birds, in: Croxall, J.P. (Ed.), Seabirds: Feeding Ecology and Role in Marine Ecosystems. Cambridge Univ Press.

Hüppop, O., Dierschke, J., Exo, K.M., Fredrich, E., Hill, R., 2006. Bird migration studies and potential collision risk with offshore wind turbines. Ibis 148, 90–109.

Jonsen, I., Myers, R., Flemming, J., 2003. Meta-analysis of animal movement using state-space models. Ecology 84, 3055–3063.

Kahlert, J., Desholm, M., Clausager, I., 2004. Investigations of migratory birds during operation of Nysted offshore wind farm at Rodsand: Preliminary analysis of data from spring 2004. -Note from NERI commisioned by Energi E2.

Kendal, W.S., 2004. Taylor's ecological power law as a consequence of scale invariant exponential dispersion models. Ecological Complexity

1, 193–209.

Kilpatrick, A., Ives, A., 2003. Species interactions can explain Taylor's power law for ecological time series. Nature 422, 65–68.

King, D.A., 2004. Climate Change Science : Adapt, Mitigate, or Ignore? Science 303, 176–177.

Kotliar, N., Wiens, J., 1990. Multiples scales of patchiness and patch structure: a hierarchical framework for the study of heterogeneity. Oikos 59, 253–260.

Kunz, T.H., Arnett, E.B., Cooper, B.M., Erickson, W.P., Larkin, R.P., Mabee, T., Morrison, M.L., Strickland, M.D., Szewczak, J.M., 2007. Assessing Impacts of Wind-Energy Development on Nocturnally Active Birds and Bats: A Guidance Document. Journal of Wildlife Management 71, 2449–2486.

Laborde, P., Urrutia, J., Valencia, V., 1999. Seasonal variability of primary production in the Cap-Ferret Canyon area (Bay of Biscay) during the ECOFER cruises. Deep-Sea Research II 46, 2057–2079.

Larsen, J., Guillemette, M., 2007. Effects of wind turbines on flight behaviour of wintering common eiders: implications for habitat use and collision risk. Journal of Applied Ecology 1, 516–522.

Leinster, T., Cobbold, C., 2012. Measuring diversity: the importance of species similarity. Ecology 93, 477–489.

Lewis, S., Sherratt, T.N., Hamer, K.C., Wanless, S., 2001. Evidence of intra-specific competition for food in a pelagic seabird. Nature 412, 816–819.

Linley, E.A.S., Wilding, T.A., Black, K., Hawkins, A.J.S., Mangi, S., 2007. Review of reef effects of offshore wind farm structures and potential for enhancement and mitigation. Report from PML Applications Ltd. and the Scottish Association for Marine Science to the Department for Business, Enterprise and Regulatory Reform (BERR).

Llope, M., Anadón, R., Viesca, L., Quevedo, M., González-Quirós, R., Stenseth, N.C., 2006. Hydrography of the southern Bay of Biscay shelf-break region: Integrating the multiscale physical variability over the period 1993–2003. Journal of Geophysical Research 111.

Louzao, M., Hyrenbach, K.D., Arcos, J., Abelló, P., de Sola, L.G., Oro, D., 2006. Oceanographic habitat of an endangered Mediterranean procellariiform: Implications for marine protected areas. Ecological Applications 16, 1683–1695.

MARM, MITYC, 2009. Estudio estratégico ambiental del litoral español para la instalación de parques eólicos marinos.

Madsen, P., Wahlberg, M., Tougaard, J., 2006. Wind turbine underwater noise and marine mammals: implications of current knowledge and data needs. Marine Ecology Progress Series 309, 279–295.

Maestro, A., López-Martínez, J., Llave, E., Bohoyo, F., Acosta, J., Hernández-Molina, F.J., Muñoz, A., Jané, G., 2012. Geomorphology of the Iberian Continental Margin. Geomorphology.

Mallory, M.L., Gilchrist, H.G., Braune, B.M., Gaston, A.J., 2006. Marine birds as indicators of Arctic marine ecosystem health: linking the Northern Ecosystem Initiative to long-term studies. Environmental monitoring and assessment 113, 31–48.

Masden, E. a, Fox, A.D., Furness, R.W., Bullman, R., Haydon, D.T., 2010. Cumulative impact assessments and bird/wind farm interactions: Developing a conceptual framework. Environmental Impact Assessment Review 30, 1–7.

Masden, E. a, Haydon, D.T., Fox, A.D., Furness, R.W., 2010. Barriers to movement: Modelling energetic costs of avoiding marine wind farms amongst breeding seabirds. Marine Pollution Bulletin 60, 1085–91.

Mateos, M., 2009. Radar technology applied to the study of seabird migration across the Strait of Gibraltar.

Mcsorley, C.A., Webb, A., Dean, B.J., Reid, J.B., 2005. UK inshore marine Special Protection Areas: a methodological evaluation of site selection and definition of an interest feature using line transect data. JNCC Report No 344.

Moreno, R., 2010. Seabirds as bioindicators of coastal ecosystem recovery following the Prestige oil spill. Thesis.

Navarro, J., Oro, D., Bertolero, A., Genovart, M., Delgado, A., Forero, M.G., 2010. Age and sexual differences in the exploitation of two anthropogenic food resources for an opportunistic seabird. Marine Biology 157, 2453–2459.

Nettleship, D.N., Duffy, D.C., 1993. Seabird populations. Elsevier Applied Science, London.

Noer, H., Christensen, T.K., Clausager, I., Petersen, I.K., 2000. Effects on birds of an offshore wind park at Horns Rev: Environmental impact assessment, NERI report. NERI report, MEE-NERI.

Observ'ER, 2011. The state of renewable energies in Europe (11th EurObserv'ER Report, 2011).

Palomera, I., 1992. Spawning of anchovy Engraulis encrasicolus in the Northwestern Mediterranean relative to hydrographic features in the region. Marine Ecology Progress Series 79, 215–223.

Patterson, T. a, Thomas, L., Wilcox, C., Ovaskainen, O., Matthiopoulos, J., 2008. State-space models of individual animal movement. Trends in ecology & evolution 23, 87–94.

Pearce, J., Ferrier, S., 2001. The practical value of modelling relative abundance of species for regional conservation planning: a case study. Biological Conservation 98, 33–43.

Peliz, Á., Dubert, J., Santos, a. M.P., Oliveira, P.B., Le Cann, B., 2005. Winter upper ocean circulation in the Western Iberian Basin—Fronts, Eddies and Poleward Flows: an overview. Deep Sea Research Part I 52, 621–646.

Percival, S.M., 2003. Birds and wind farms in Ireland: A review of potential issues and impact assessment. Report comissioned by the Sustainable Energy Authority of Ireland.

Perrow, M.R., Gilroy, J.J., Skeate, E.R., Tomlinson, M.L., 2011. Effects of the construction of Scroby Sands offshore wind farm on the prey base of Little tern Sternula albifrons at its most important UK colony. Marine Pollution Bulletin 62, 1661–70.

Perrow, M.R., Skeate, E.R., Lines, P., Brown, D., Tomlinson, M.L., 2006. Radio telemetry as a tool for impact assessment of wind farms: the case of Little Terns Sterna albifrons at Scroby Sands, Norfolk, UK. Ibis 148, 57–75.

Piatt, J.F., Sydeman, W.J., 2007. Seabirds as indicators of marine ecosystems. Marine Ecology Progress Series 352, 199–204.

Planque, B., Lazure, P., Jégou, A., 2004. Detecting hydrological landscapes over the Bay of Biscay continental shelf in spring. Climate research 28, 41–52.

Punt, M.J., Groeneveld, R. a., van Ierland, E.C., Stel, J.H., 2009. Spatial planning of offshore wind farms: A windfall to marine environmental protection? Ecological Economics 69, 93–103.

Ribas-Ribas, M., Gómez-Parra, A., Forja, J.M., 2011. Spatio-temporal variability of the dissolved organic carbon and nitrogen in a coastal area affected by river input: The north eastern shelf of the Gulf of Cádiz (SW Iberian Peninsula). Marine Chemistry 126, 295–308.

Rosenzweig, C., Karoly, D., Vicarelli, M., Neofotis, P., Wu, Q., Casassa, G., Menzel, A., Root, T.L., Estrella, N., Seguin, B., Tryjanowski, P., Liu, C., Rawlins, S., Imeson, A., 2008. Attributing physical and biological impacts to anthropogenic climate change. Nature 453, 353–357.

Salat, J., 1996. Review of hydrographic environmental factors that may influence anchovy habitats in northwestern Mediterranean. Scientia Marina.

Santos, A.M.P., Chícharo, A., Dos Santos, A., Moita, T., Oliveira, P.B., Peliz, Á., Ré, P., 2007. Physical–biological interactions in the life history of small pelagic fish in the Western Iberia Upwelling Ecosystem. Progress In Oceanography 74, 192–209.

Schwemmer, P., Garthe, S., 2008. Regular habitat switch as an important feeding strategy of an opportunistic seabird species at the interface between land and sea. Estuarine, Coastal and Shelf Science 77, 12–22.

Skeate, E.R., Perrow, M.R., Gilroy, J.J., 2012. Likely effects of construction of Scroby Sands offshore wind farm on a mixed population of harbour Phoca vitulina and grey Halichoerus grypus seals. Marine pollution bulletin.

Spear, L., Ainley, D., Hardesty, B., 2004. Reducing biases affecting at-sea surveys of seabirds: use of multiple observer teams. Marine Ornithology 157, 147–157.

Tasker, M.L., Jones, P.H., Dixon, T. m, Blake, B.F., 1984. Counting seabirds at sea from ships: a review of methods employed and a suggestion for a standardized approach. The Auk 101, 567–577.

Taylor, L.R., 1961. Aggregation, Variance and the Mean. Nature 189, 732–735.

Taylor, L.R., Woiwod, I.P., 1980. Temporal Stability as a Density-Dependent Species Characteristic. The Journal of Animal Ecology 49, 209–224.

Taylor, L.R., Woiwod, I.P., 1982. Comparative Synoptic Dynamics. I. Relationships between Inter- and Intra-specific spatial and temporal variance/mean population parameters. The Journal of Animal Ecology 51, 879–906.

Taylor, L.R., Woiwod, I.P., Perry, J.N., 1980. Variance and the Large Scale Spatial Stability of Aphids, Moths and Birds. The Journal of Animal Ecology 49, 831–854.

Tobin, P.C., 2004. Estimation of the spatial autocorrelation function: consequences of sampling dynamic populations in space and time. Ecography 27, 767–775.

Tremblay, Y., Bertrand, S., Henry, R., Kappes, M., Costa, D., Shaffer, S., 2009. Analytical approaches to investigating seabird–environment interactions: a review. Marine Ecology Progress Series 391, 153–163.

Turchin, P., 1998. Quantitative Analysis of Movement: Measuring and Modeling Population Redistribution in Animals and Plants. Sinauer.

Whitehouse, R.J.S., Harris, J.M., Sutherland, J., Rees, J., 2010. The nature of scour development and scour protection at offshore windfarm foundations. Marine Pollution Bulletin 62, 73–88.

Wiese, F.K., Montevecchi, W.., Davoren, G.., Huettmann, F., Diamond, A.., Linke, J., 2001. Seabirds at risk around offshore oil platforms in the North-west Atlantic. Marine Pollution Bulletin 42, 1285–1290.

Wu, J., David, J.L., 2002. A spatially explicit hierarchical approach to modeling complex ecological systems: theory and applications. Ecological Modelling 153, 7–26.

Catalan summary

INTRODUCCIÓ

INTRODUCCIÓ

La creixent demanda mundial d’energia i el canvi climàtic són dos dels grans desafiaments d’aquest segle. En aquest escenari és necessari trobar un equilibri entre les polítiques de canvi climàtic i la competitivitat, per tal que la reducció les emissions de carboni sigui econòmicament viable. En aquest context, la Comissió Europea va definir el «full de ruta Energia 2050», que explora les possibilitats d›aconseguir una economia baixa en carboni i que alhora s›assegura un subministrament d›energia competitiva, sostenible i segura (CE, 2011). La Unió Europea s›ha compromès a reduir les emissions de gasos d›efecte hivernacle fins a un 80-95% per sota dels nivells de 1990 per al 2050 (CE, 2011b). És impossible predir els canvis que es produiran a Europa a llarg termini, però alguns dels possibles escenaris són: (i) un sistema d’alta eficiència energètica, (ii) un sistema amb una oferta diversificada de tecnologia, com ara la captura de carboni i les instal·lacions d›emmagatzematge o l’energia nuclear, i (iii) un fort suport a les fonts d›energia renovables. Tots els pronòstics per reduir les emissions de carboni a Europa mostren que l›electricitat haurà de tenir un paper més important que els combustibles fòssils i que la participació de les fonts d›energia renovables s›incrementarà substancialment, fins a assolir un 55%, 64 o 97% del consum d›energia final bruta el 2050, segons l›escenari (CE, 2011b). Una de les mesures polítiques per aconseguir aquest objectiu és la Directiva sobre energies renovables, que fixa com a objectiu que el 20% del consum energètic provingui de fonts renovables el 2020.

A Europa, les energies renovables representen el 18% de tota la producció energètica (Eurostat, 2009;. Fig 1a). L›energia hidroelèctrica és la font principal de producció d’energies renovables (54,5%), seguida de l›energia eòlica (22,5%) (Observ›ER de 2011,. Fig 1b). L’any 2050 s’espera que l›energia eòlica proporcioni més electricitat que qualsevol altra tecnologia (CE, 2011b) i per tant la contribució potencial del medi marí per al desenvolupament d›energia eòlica marina ha rebut una gran atenció a les últimes dècades.

El primer parc eòlic marí es va instal lar a Dinamarca el 1991. Des de llavors, el sector ha tingut una ràpida expansió, particularment al nord d›Europa. Avui en dia, Europa és el líder mundial en energia eòlica marina, amb un total de 1.371 turbines a alta mar distribuïdes en 53 parcs eòlics en 10 països a finals del 2011 (EWEA, 2012). El Regne Unit és el país amb la major capacitat instal·lada d›energia eòlica marina, seguit per Dinamarca, Països Baixos i Alemanya. L›interès per l›energia eòlica marina s›està estenent més enllà d›Europa. A Xina, Japó, Corea del Sud, EUA i Israel hi ha empreses que treballen en el desenvolupament de turbines eòliques al mar, encara que només la Xina té tres parcs eòlics marins operatius.

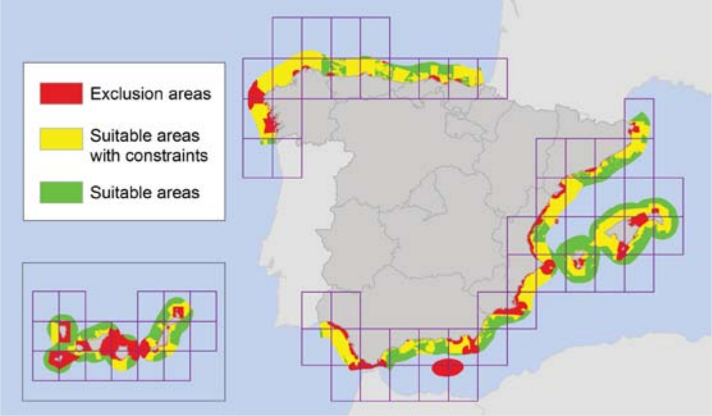
La majoria de les turbines instal·lades fonaments al fons marí. S'estan desenvolupant models flotants, i Noruega i Portugal són els primers països que tenen instal·lada una turbina flotant a gran escala. A mesura que la tecnologia maduri, s'espera que els parcs eòlics marins creixin en grandària i que es despleguin més lluny de la costa i

Quadre *Estat de desenvolupament de l'energia eòlica marina a Espanya*

ENERGIA EÒLICA MARINA A ESPANYA

Actualment Espanya no té parcs eòlics marins operatius. Des del començament de l›expansió de l›energia eòlica en alta mar al nord d›Europa, diferents empreses han mostrat el seu interès en la construcció de parcs eòlics marins a les costes espanyoles. Malgrat les iniciatives del sector privat per promoure el seu desenvolupament, el govern espanyol va establir l'any 2007 el procediment administratiu obligatori per aconseguir la concessió per construir un parc eòlic marí a les costes espanyoles (Reial Decret 1028/2007).

El procediment establia com a necessari una Avaluació Ambiental Estratègica (AAE) de la costa espanyola. Aquest estudi va ser publicat el 2009 (MARM i el MITYC, 2009) i va incloure el mapa de zonificació definitiva per a les àrees de desenvolupament de l'energia eòlica. Aquest mapa divideix les costes espanyoles en 72 àrees (definits per un grau quadrats decimals). Les primeres 24 milles nàutiques de cada àrea es van avaluar d›acord a múltiples criteris per a classificar les àrees en tres categories: aptes (en verd), àrees adequades amb restriccions (en groc) i les zones d›exclusió (en vermell).



El procés de concessió administrativa és llarg i complex i ha patit diversos retards. Avui en dia no hi ha un nombre oficial dels parcs eòlics previstos a Espanya.

en aigües més profundes, sobretot si se’n demostra la viabilitat econòmica. Els projectes actuals en construcció tenen una profunditat mitjana de 25 metres i una distància de la costa de 33 km (EWEA, 2012), ja que molts dels OWF s›han construït al Mar del Nord, que té una gran part a la plataforma continental europea. Això proporciona regions planes i superfícies relativament grans adequades per al desenvolupament d’OWF (Henderson et al., 2003). A diferència del nord d›Europa, la costa oest de França, la Península Ibèrica i la Mediterrània segueixen sent un desafiament per al desenvolupament d’OWF. Encara que hi ha projectes previstos per a aquestes zones, les turbines disponibles i els mètodes de fonamentació requeririen la construcció de parcs eòlics molt més a prop de la línia de costa, amb un consegüent increment dels conflictes per trobar llocs òptims, ja sigui per l’acceptació social, els impactes ambientals, els conflictes de interès nacional o la planificació espacial marina. Tots aquests factors, juntament amb la manca de finançament, estan frenant el desenvolupament de l›energia eòlica en alta mar a l›oest i sud d›Europa.

De fet, l’energia eòlica marina no està exempta de conflictes. A escala global, el canvi cap a les energies renovables és acceptat àmpliament com un pas necessari per mitigar els efectes del canvi climàtic antropogènic (King, 2004;. Rosenzweig et al, 2008). A escala local, però, cal considerar acuradament els impactes ambientals del desenvolupament de l›energia eòlica (Gill, 2005). En el camp de la gestió del medi marí, hi ha una creixent preocupació sobre el desenvolupament de l›energia eòlica al mar i els seus possibles impactes en l›ecosistema marí. Alguns dels aspectes que s›estan estudiant són l›alteració del fons i fauna marines durant la construcció i operació OWF (Whitehouse et al, 2010; .. Burkhard et al, 2011) i els efectes sobre les larves de peixos (Perrow et al, 2011). A més, es desconeixen els efectes del soroll submarí sobre la vida dels peixos i mamífers marins (Madsen et al, 2006; .. Bailey et al, 2010), els efectes a nivell de la població de les col·lisions d›aus amb les turbines (Fox et al, 2006;. Desholm , 2009) i els efectes de la pertorbació (Drewitt i Langston, 2006;. Masden, Haydon, et al, 2010).

L’AVALUACIÓ AMBIENTAL

La Unió Europea compta amb un marc normatiu (Directiva 2001/42/CE) per estandarditzar l’avaluació i el seguiment de les activitats humanes en els ecosistemes i garantir un desenvolupament racional d’aquestes activitats, incloent consideracions ambientals. A gran escala, els països han de desenvolupar una Avaluació Ambiental Estratègica (AAE) per planificar la seva xarxa de parcs eòlics marins i minimitzar el seu impacte ecològic sobre el medi ambient costaner. A nivell local, cada projecte de parc eòlic requereix una Avaluació d›Impacte Ambiental (EIA) dels possibles impactes negatius del projecte proposat en el medi marí.

Durant molts anys, l›única informació disponible sobre els parcs eòlics en alta mar eren informes es van centrar en la forma de realitzar les EIA de projectes particulars. L›experiència danesa amb els primers parcs eòlics i la seva metodologia d›aixecament aeri s›ha convertit en un referent per a molts (EIA Noer et al., 2000). Més tard, el COWRIE (Collaborative Offshore Wind Research Into the Environment) del Regne Unit va encarregar un informe per estandarditzar les tècniques de censos d’aus marines per l’EIA de parcs eòlics a alta mar (Camphuysen et al., 2004). En els últims anys, i com que el sector ha crescut, s’han publicat reportatges i treballs d›investigació sobre l›avaluació de la interacció amb el medi ambient de parcs eòlics particulars (per exemple Desholm i Kahlert, 2005; Perrow et al, 2011;. Skeate et al, 2012), i també revisions i treballs generals relacionades amb l›EAE (Elliott, 2002;. Fox et al, 2006;. Punt et al, 2009; .. Masden, Fox, et al, 2010).

LES AUS MARINES COM INDICADORS

Els ecosistemes marins tenen nivells de biodiversitat elevats i a la vegada són molt complexos ecològicament. Mentre que els estudis ecològics es centren en aquesta complexitat, l'ecologia aplicada requereix de mètodes que sintetitzen aquesta complexitat per tal de prendre mesures que puguin tenir conseqüències econòmiques (Platt i Sydeman, 2007). Aquest és el cas de la utilització d›espècies indicadores per simplificar els processos de supervisió i de gestió per a les EIA i EAE. Els principals depredadors marins són un component clau de la gestió dels ecosistemes marins (Boyd et al, 2006.), i dins dels principals depredadors, les aus marines s›han convertit en els indicadors generalitzats per avaluar els efectes potencials de les activitats humanes al mar, així com la salut de l›ecosistema (Cairns, 1987; Nettleship i Duffy, 1993; Mallory et al, 2006).

Les aus marines ofereixen molts avantatges en comparació amb altres espècies. Considerant un entorn on la majoria de les espècies estan sota l›aigua, les aus marines són animals visibles que es poden estudiar fàcilment. A més, com que algunes espècies són fàcils de capturar, es poden realitzar seguiments individuals i estudis demogràfics (Platt i Sydeman, 2007). D›altra banda, la majoria de les aus marines tenen determinats marcs legals de protecció (com ara la Directiva Aus i la Directiva Hàbitats a Europa) i són espècies emblemàtiques per al públic (Fox et al., 2006). Per aquests motius hi ha una gran abundància d›amplis estudis a llarg termini de la seva distribució en el mar i les tendències poblacionals.

Per tot això, la distribució i abundància de les aus marines esdevé informació clau per donar suport a àrees marines protegides (Garthe et al, 2011; .. Arcs et al, 2012), per aplicar les mesures de gestió de la pesca (Boyd et .. al, 2006), per monitoritzar l›impacte de les plataformes de petroli i gas al mar (Wiese et al, 2001), o per avaluar l›impacte dels desastres ambientals com ara vessaments de petroli (Bretagnolle et al, 2004;. Moreno, 2010) . Per tant, les aus marines són indicadors adequats del medi marí, i s›han convertit en una de les pedres angulars del procés de presa de decisions per a la selecció d›àrees òptimes per al desenvolupament nacional d›energia eòlica marina i l›avaluació d›impacte dels projectes de OWF particulars.

Els impactes potencials sobre les aus marines

A l'hora de seleccionar les àrees de desenvolupament, o quan la ubicació d'un projecte es resol, podem diferenciar l'efecte de OWF en dos tipus d'aus: i) espècies migratòries que poden trobar els parcs eòlics en les seves rutes migratòries i ii) espècies que tenen la seva zona de cria i hivernada prop d'on es situa el parc eòlic. Tots dos tipus d›aus són susceptibles a múltiples impactes antropogènics (Anderson et al, 2003;. Hüppop et al, 2006; .. Louzao et al, 2006), però els impactes potencials dels parcs eòlics marins sobre les comunitats d›aus marines es poden classificar en tres tipus, (i) mortalitat directa a través de la col lisió, (ii) modificació del seu hàbitat físic i (iii) efectes de pertorbació i de barrera.

Risc de col·lisió

Els ocells que volen a la zona del parc

eòlic tenen clarament un cert risc de col·lisió amb les aspes i l›estructura estacionària, o de ser atrapats i ferits en els vòrtex de pressió creats per les pales del rotor (Fox et al., 2006). El risc de col·lisió depèn d›una sèrie de factors relacionats amb les espècies d›aus (maniobrabilitat, envergadura, etc), comportament (per exemple, activitat nocturna), la presència en grans quantitats i les condicions meteorològiques que redueixen la visibilitat. La mortalitat per col·lisió és el perill més important, ja que la mortalitat directa pot tenir potencialment conseqüències ràpides en els nivells de població. No obstant això, encara hi ha poca informació sobre el nombre real de col·lisions d›aus amb parcs eòlics en alta mar, en gran part com a conseqüència de les dificultats tècniques per detectar les col·lisions al mar (Drewitt i Langston, 2006).

Modificació de l'hàbitat

Aquest impacte comprèn la pèrdua d›hàbitat que resulta de la presència de les bases de turbina, els cables de connexió a la xarxa i qualsevol altra construcció associada. La magnitud de la pèrdua d›hàbitat no es considera generalment com una de les principals preocupacions quan això no es produeix en zones d›alta biodiversitat o importància ecològica (BirdLife International, 2003). No obstant això, també pot haver-hi una pèrdua d›hàbitat indirecta a causa dels fonaments de la turbina sobre el fons del mar, o pel canvi en l›ús de l›hàbitat que en fan els humans. Per exemple, l›activitat de la construcció i la distribució de la turbina pot afectar la hidrologia del lloc i tenir un impacte al llarg de grans àrees (Percival, 2003). Hi ha incertesa sobre la magnitud d›aquests canvis, però el dany pot ser significatiu, especialment en les àrees d›alimentació, com ara bancs de sorra en aigües poc profundes (Drewitt i Langston, 2006).

Les bases de les turbines tendeixen a tenir un «efecte escull» que augmenta la biodiversitat a través de la creació d›hàbitat (Linley et al., 2007), però això pot influir en les comunitats de flora i fauna de manera complexa generant efectes tant positius com negatius, depenent del lloc i de l›espècie (Perrow et al., 2011). Les aus marines també poden veure's afectades de manera diferent pels canvis en l›hàbitat. Mentre que algunes espècies especialistes poden perdre importants fonts d›aliment, altres espècies oportunistes (com ara les gavines) poden augmentar la seva presència a la zona per explotar la nova font d›aliment. D'altres aus marines (com succeeix amb els cormorans) poden veure's atretes per les plataformes de manteniment de turbines que utilitzen com estructures de descans (Kahlert i col., 2004). No obstant això, aquest guany d›hàbitat podria ser contrarestat per un risc de col·lisió superior.

Pertorbació

La presència de les turbines, així com els moviments dels vaixells i de les persones relacionades amb la construcció i manteniment del lloc, pot dissuadir algunes aus marines de l'ús de zones del parc eòlic i els seus voltants. L›escala dels efectes de pertorbació varia molt en funció d›una àmplia gamma de factors (Drewitt i Langston, 2006), com ara el disseny de la matriu de la turbina i la distància entre les turbines; els patrons d'activitat (nocturna o diürna) de les aus marines (Desholm i Kahlert, 2005); o les condicions climàtiques. Les respostes conductuals als parcs eòlics no només poden variar entre les espècies, sinó també entre individus de la mateixa espècie en funció de factors com ara l›etapa del cicle de vida (hivernada, muda i de cria), la mida o la tendència a l›habituació.

Encara que es produeixi pertorbació i desplaçament, el seu efecte pot ser intranscendent si hi ha abundància d›hàbitats alternatius. No obstant això, els parcs eòlics marins situats a les rutes migratòries o en trajectòries de vol locals podrien alterar els moviments de les aus i augmentar la seva despesa d›energia (Masden, Haydon, et al., 2010). Aquest tipus de trastorn s›anomena «efecte barrera». De fet, les observacions en parcs eòlics operatius mostren que molts ocells decideixen volar fora del parc eòlic en lloc de volar entre les turbines (Desholm i Kahlert, 2005; Larsen i Guillemette, 2007). Desafortunadament, hi ha una manca d›informació completa abans i després de l›impacte (BACI) en molts parcs eòlics operatius per quantificar adequadament la barrera i els efectes de les pertorbacions en comparació amb el comportament bàsic de les aus marines (Drewitt i Langston, 2006).

Noves línies de recerca

Les directrius acordades internacionalment recomanen l'avaluació del risc de col·lisió amb estudis de radar en àrees fortament migratòries (Desholm et al, 2006;. Fox et al, 2006; Kunz et al, 2007) i mapes de densitat per avaluar la pèrdua de hàbitats d'alimentació i la modificació de l›hàbitat físic (Camphuysen et al, 2004;. Fox i Petersen, 2006).

Per a avaluar el risc de col·lisió la tecnologia de radar és una eina poderosa, ja que permet millorar el nostre coneixement sobre patrons espaciotemporals d›alguns grups d›aus marines. La recopilació de dades de radars i l›anàlisi dels resultats requereixen estudis integrals que ja s›han abordat (Desholm, 2006; Brookes, 2009; Mateos, 2009). Per contra, l›ús de mapes de densitat s›ha quedat enrere en la integració de la dimensió espaciotemporal dels patrons de les aus marines, tot i que els mapes de distribució de les aus marines tenen un paper prominent en la majoria d›EIA i les avaluacions dels mars. Respecte a la distribució d›aus marines i la seva abundància, generalment es dóna com a simples localitzacions o quadrícules de densitat. Després d›una revisió de més de 200 estudis publicats, Tremblay et al. (2009) va assenyalar que «la visualització senzilla de les dades de distribució ha estat molt més freqüent que els índexs quantitatius». De fet, pocs estudis han tractat d›abordar els mètodes analítics i sintètics per extreure les decisions adequades estratègiques (AAE) o locals (EIA) dels nivells de les dades de distribució d›aus marines. Aquesta tesi pretén contribuir a omplir aquest buit en l›enfocament metodològic per a l›ús de les dades de distribució d›aus marines en alta mar per les Avaluacions d›Energia Eòlica.

OBJECTIUS

OBJECTIU PRINCIPAL

L’objectiu principal d’aquesta tesi és aprofundir en les eines analítiques en l’espai i el temps per a l›avaluació ambiental de l’energia eòlica marina a fi de proporcionar als professionals les directrius sobre com i quan aplicar-les.

OBJECTIUS ESPECÍFICS

Per aconseguir aquest objectiu, la present tesi s’ha estructurat en quatre capítols i una anàlisi global que aborden els següents objectius específics:

1. Dissenyar i posar a prova un índex de vulnerabilitat per avaluar els efectes potencials de desenvolupament d’energia eòlica marina a les aus marines. (Capítols 1 i 2)
2. Desenvolupar una eina per integrar la variabilitat espacial i temporal de l›abundància d›aus marines al mar per quantificar els impactes potencials dels parcs eòlics marins a les aus marines. (Capítol 3)
3. Demostrar les limitacions dels mapes de distribució i abundància a través del seguiment basat en individus d›una espècie emblemàtica. (Capítol 4)
4. Proporcionar directrius pràctiques sobre la manera d›integrar les eines analítiques presentades en el disseny d›EAE i EIA. (Discussió)

PLANTEJAMENT METODOLÒGIC

CENSOS D’AUS MARINES

De les tècniques de cens existents, els millors mètodes disponibles per a la obtenció de la distribució i abundància d’aus al mar són censos en aeronau i en embarcació. Els censos en vaixell han estat àmpliament utilitzats seguint una metodologia estandarditzada (Tasker et al., 1984), amb adaptacions d›acord a cada projecte en particular. Els censos aeris d›aus marines al mar han tingut una ràpida expansió en l›última dècada, que ha estat fortament influenciada per l›experiència danesa en relació amb l›avaluació de l›impacte ambiental dels parcs eòlics a alta mar (Camphuysen et al., 2004). Fins ara, el mètode exposat en els seus informes (per exemple, Noer et al., 2000) s›ha convertit en un referent.

L›elecció entre un o altre mètode depèn de la topografia i dels objectius específics de la investigació, ja que cada mètode té els seus avantatges i desavantatges (Camphuysen et al., 2004 per a una revisió completa). Els censos en vaixell són especialment adequats per fer recomptes exhaustius, el que permet una millor identificació de les espècies amb el temps suficient per recollir informació addicional com l›edat, el comportament o l›alçada de vol. No obstant això, aquest mètode té dos desavantatges principals. En primer lloc, els vaixells al mar, encara que no proporcionin aliments, tenen un efecte d›atracció sobre les aus que modifica en algun grau la distribució original de les aus marines (Spear et al., 2004). En segon lloc, aquest mètode requereix un temps més llarg al mar per cobrir grans àrees. Els estudis aeris, d›altra banda, són particularment eficaços en una cobertura simultània de grans àrees que proporcionen una instantània de distribució i abundància (Camphuysen et al., 2004) amb un mínim d›atracció o repulsió (Certain i Bretagnolle, 2008). D›altra banda, els reconeixements aeris permeten estudiar zones llunyanes de difícil accés (com ara, zones poc profundes o bancs de sorra) en intervals de temps curts i de manera poc costosa (Camphuysen et al, 2004; .. Garthe et al, 2011). Això és possible gràcies a la velocitat dels avions, però aquesta velocitat és també la principal desavantatge del mètode. Els estudis aeris es duen a terme a la velocitat mínima de vol, que garanteix la seguretat de vol i proporciona suficient temps d›observació (en general 185 kmh). A aquesta velocitat, el temps d’observació és curt i això porta a problemes d›identificació d›algunes espècies, de precisió i reducció de la capacitat de detecció d›espècies rares i petites que són difícils de detectar a partir de l›aeronau (Camphuysen et al, 2004; .. Henkel et al, 2007). A més, la informació addicional no sempre és fàcil de recollir i no es pot calcular l’alçada de vol.

En aquesta tesi s›han utilitzat censos des de vaixell i reconeixements aeris com a font de dades de distribució d›aus marines. Les aus marines presenten patrons dinàmics dependents de l›escala de distribució, per això calen conjunts de dades que permetin fer front a aquesta variabilitat i que es puguin repetir fàcilment en condicions similars. Els estudis aeris permeten obtenir dades d’una àrea en particular diverses vegades dins d›un any (capítol primer i tercer). Els censos des de vaixells requereixen més temps però maximitzen la riquesa d›espècies detectades (nombre d›espècies o tàxons identificats en cada enquesta) (Henkel et al., 2007), una característica clau per capturar els patrons detallats de biodiversitat. En el segon capítol, l›àrea d›estudi abasta les costes de la Península Ibèrica. Censos repetits simultània i sistemàticament no eren econòmicament viables. Per tant, la maximització de la detecció de les espècies a través de censos amb vaixell era particularment important.

Tots dos tipus de censos permeten observar una àrea determinada per veure si les aus l›utilitzen, però el que sembla més intuïtiu és controlar les aus marines per estudiar com estan utilitzant una àrea (Perrow et al., 2006). Per això s›han utilitzat mitjans electrònics de seguiment, com ara transmissors de localització per satèl·lit, receptors GPS o ràdio telemetria. Des de principis de 1990, la utilització de la telemetria ha augmentat constantment a causa dels avenços en la miniaturització dels dispositius electrònics (Tremblay et al., 2009).

En l›avaluació de la interacció de l›energia eòlica en alta mar amb les aus marines, la telemetria esdevé un mètode eficaç per integrar la dimensió espacial i temporal dels patrons de distribució de les aus marines. No obstant això, hi ha alguns inconvenients per a aquesta metodologia. Alguns d›aquests dispositius tenen alts costos; la grandària mostral és petita, i per tant cal una gran quantitat de temps d›anàlisi. A més només un nombre limitat d›espècies d›aus marines poden ser capturades per fixar els mètodes de marcatge (Perrow et al, 2006.). Aquest enfocament, que s’ha utilitzat en el quart capítol de la tesi, proporciona estudis de comportament a escala fina i resulta especialment útil si s›utilitza juntament amb mètodes com censos aeris i des de vaixell (Tremblay et al., 2009).

LES ÀREES D’ESTUDI

Aquesta tesi doctoral aborda la qüestió del desenvolupament d’energia eòlica marina i les interaccions de les aus, des d’una perspectiva metodològica, sense centrar-se en una àrea particular. No obstant això, per tal de presentar un instrument d›anàlisi, entendre l›eina, la seva aplicació i aplicabilitat per a la presa de decisions i la gestió, les dades reals són molt millor que els conjunts de dades simulades. Les tres àrees d›estudi pertanyen a les aigües franceses, portugueses i espanyoles i tenen un gran potencial per al futur desenvolupament d›energia eòlica marina. A excepció de la turbina experimental flotant a Portugal, fins ara no hi ha cap OWF construït a les àrees d›estudi, el que els fa exemples rellevants de com aplicar les eines d›anàlisi per a la futura presa de decisions. A continuació s’exposa una breu descripció de les tres àrees.

Golf de Biscaia

El Golf de Biscaia és un golf de l'Oceà Atlàntic que s'estén entre el cap Ortegal a Galícia, Espanya (43.77 ° N, 7.89 ° W) i l'illa d'Ouessant, a Bretanya, França (48.43 ° N, 18/05 ° W). Dins d›aquesta zona, una regió de 100.000 km 2 es va cobrir amb 5000 km lineals de transectes aeris mensuals des d›octubre de 2001 a març de 2002 i amb 4000 km lineals de transectes des de vaixell a la primavera del 2003 al 2006.

L›àrea d›estudi cobreix la plataforma continental francesa del Golf de Biscaia entre Penmarch al nord (47.75 ° N, 28/04 ° W) i Baiona al sud (43.497 ° N, 1.64 ° W). Les àrees de descans costaneres i de la plataforma són els sistemes més productius de la regió (Certain et al., 2008). Les desembocadures dels rius Loira i Gironde són una font d›aigua rica en nutrients frescs (Planque et al., 2004) i la vora de la plataforma és una àrea d'elevada producció primària, on les aigües profundes més fredes arriben a la capa eufòtica, a causa de les marees internes i les onades (Gerkema et al., 2004), especialment a la zona sud, que es caracteritza per un profund canó, el Cap Ferret (Laborde et al., 1999).

La comunitat d›aus marines d'aquesta àrea es pot classificar en vuit famílies i un total de 30 espècies.

Costes de la península ibèrica

Aquesta àrea d'aproximadament 230.000 km 2, cobreix la plataforma continental espanyola i portuguesa i s'estén sobre 7.800 km de costa. Els censos es van dur a terme en vaixell per SPEA (la Societat Portuguesa per a l'Estudi de les Aus) i SEO / Birdlife (Societat Espanyola d'Ornitologia) en diferents etapes entre 1999 i 2011.

La ubicació de la Península Ibèrica, envoltada per l'Oceà Atlàntic i el Mar Mediterrani, i la diversitat geomorfològica i oceanogràfica dels seus marges continentals, té implicacions significatives sobre el seu clima i la circulació de masses d'aigua (Mestre et al., 2012 per a una revisió detallada). El marge continental de la Península Ibèrica té diverses regions ben diferenciades diverses regions: i) la regió est de l'Atlàntic Nord i l›aflorament ibèric cap al pol actual, que tenen una forta influència en els marges continentals portuguès, gallec i del Golf de Biscaia (Peliz et al., 2005; Llope et al, 2006); ii) la sortida de l›aigua del Mediterrani que flueix des de l›estret de Gibraltar al llarg del talús continental del Golf de Cadis (Ribas-Ribas et al, 2011); iii) els corrents de l›Atlàntic que afecten el Mar d›Alborán , i iv) altres masses d›aigua mediterrànies que influeixen en els marges continentals valència, el català i balear (Salat, 1996). Aquesta configuració oceanogràfica afecta la composició i l›estructura del plàncton i de tots els components de la cadena alimentària (Sants et al, 2007; .. Cabal et al, 2008) fins als nivells tròfics més alts i, per tant, les aus marines. De fet, la Península Ibèrica alberga la major diversitat d›aus marines d'Europa. La comunitat d›aus marines en aquesta zona té fins a 39 espècies habituals, a més d›espècies rares que amb el temps es poden trobar.

Delta de l'Ebre

La tercera àrea, d'escala més local, es troba al voltant de Delta de l'Ebre (40,7 ° N, 0.75 °E) L›àrea d›estudi cobreix 1.435 km 2 de la plataforma continental des del port de l'Ametlla de Mar (24 km al nord; 40,86 ° N, 0,8 ° E) fins a Peñíscola (51 km al sud; 40,35 ° N, 0,4 ° E). D'aquest àrea, que pot ser coberta en un sol dia amb una aeronau, se'n van fer reconeixements aeris un cop al mes d›abril de 2005 a març de 2006.

Aquesta zona compta amb un aflorament permanent gràcies a la combinació de la influència del front liguro-provençal-català, la sobtada ampliació de la plataforma continental i la font de nutrients del riu Ebre (Palomera, 1992; Arcs, 2001). L›alta productivitat de la zona és compatible amb una important flota pesquera, que és una font d›alimentació clau per a la cria i hivernada d›aus marines al Delta de l›Ebre (Arcs, 2001;. Arcs et al, 2008). D›altra banda, el Delta de l›Ebre és una zona humida d›importància internacional inclosa en el Conveni de Ramsar des de 1993. Amb 320 km 2, és la segona zona humida més important de la Mediterrània occidental després de la Camarga a França i la segona més important de la Península Ibèrica després de Doñana. Els arrossars, llacunes, salines i platges del delta de l›Ebre ofereixen una varietat d›hàbitats de cria i hivernada de les aus, però també un punt de parada per a un gran nombre d›aus migratòries. En global, s'hi poden trobar més de 300 espècies d›aus (Bigas, 2012), 18 de les quals es van poder detectar al mar des de la aeronau.

EINES DE MODEL·LITZACIÓ

L'eficàcia de la utilització de dades sobre la distribució de les aus marines al mar com a eina per a la conservació i valoració del medi ambient depèn de si les dades espacials a partir dels censos d›aus marines representen un patró general o només una puntual «instantània» d›un sistema altament dinàmic (Fauchald et al., 2002).

Malgrat la seva homogeneïtat superficial, el mar és un entorn heterogeni a causa de les seves múltiples característiques hidrogràfiques i la distribució desigual de la seva biota (González-Solís i Shaffer, 2009). La distribució espacial i temporal dels animals és el resultat de la combinació de processos extrínsecs, relacionats amb la influència dels factors ambientals biòtics i abiòtics, i processos intrínsecs, relacionats amb la dinàmica de la població i de les interaccions intra- específiques (Bellier et al., 2010). A mes, també depèn de l'escala d'estudi.

Així, en un sistema de d'agrupacions jeràrquiques dinàmiques (Kotliar i Wiens, 1990; Allen i Hoekstra, 1991; Wu i David, 2002), els patrons a gran escala seran més estables i predictibles a causa d›una alta correlació amb les variables ambientals que defineixen els hàbitats potencials (Hunt i Schneider, 1987;. Bellier et al, 2010). En canvi, a escales espacials més petites, els patrons són menys predictibles ja que depenen de combinacions particulars de variables circumstancials que creen un hàbitat temporal preferencial dins l'hàbitat potencial (Bellier et al., 2010).

Per traduir aquests conceptes teòrics de l'ecologia aplicada calen els instruments que permetin avaluar de forma òptima les interaccions entre aus marines i OWF. Aquests hauran de tenir en compte l'efecte diferencial de les escales espacials i temporals. En avaluacions a gran escala, es pot considerar que els patrons de distribució observats són estables en el temps i representen els hàbitats potencials. Per tant, permeten delimitar de forma òptima les àrees clau de protecció (per exemple, Important Bird Areas, IBAs) i les àrees clau per al desenvolupament d›energia eòlica marina. En les avaluacions a escala regionals o local, cal avaluar l›agrupació observada d›aus marines en la seva variabilitat temporal i espacial per quantificar (amb probabilitats) el risk d'exposició a a l'OWF.

Gran escala: Índex de Sensibilitat

L'Avaluació Ambiental Estratègica integra dades a escales molt grans, de manera que podem assumir que l'escala temporal no és una prioritat i podem combinar les dades de diferents anys o períodes. Les distribucions d'aus marines poden tenir patrons diferents depenent de l'etapa del cicle de vida (hivernada, migració i reproducció), però s'espera que la seva distribució mundial sigui espacial i temporalment predictible (Fauchald et al., 2002). En altres paraules, a nivell estratègic la principal preocupació pel que fa a l'avaluació de la interacció entre les aus marines i els OWF és la superposició espacial de la distribució de les aus marines amb el desenvolupament d'àrees clau d'OWF. Això generalment es tracta amb la selecció de mapes de presència/absència d'una espècie emblemàtica o altament vulnerable als OWFs i mapes generals de densitat amb les xifres globals dels recomptes d'aus marines al mar. En aquest context, és convenient aplicar un índex per integrar i resumir totes aquestes capes d›informació.

Garthe i Hüppop (2004) van proposar l'Índex de Sensibilitat de parcs eòlics (WSI) per mapejar la vulnerabilitat de les aus marines de parcs eòlics marins a la regió del mar. Aquest índex estima primer la vulnerabilitat de cada espècie en funció de la seva sensibilitat als riscos de col·lisió i disturbis, i en funció de la seva demografia i el seu estat de conservació. Aquest valor es combina amb l›abundància espacial de cada espècie per obtenir un mapa de vulnerabilitat.

Aquest mètode és general, simple i d›àmplia aplicació. Per tant, en lloc de desenvolupar un nou índex, en aquesta tesi s›analitza el mètode en profunditat i es suggereix un refinament de la seva formulació matemàtica (capítol 1). A més, es formulen recomanacions sobre l›aplicació òptima de l›índex per a la seva utilització en qualsevol avaluació ambiental estratègica (Capítol 2).

Escala regional i local

A escales més petites, els estudis d'impacte ambiental es centren generalment en l'ús de l'hàbitat per part de les aus marines, així com en les estratègies i els processos que poden influir en l'ocurrència d'aus marines o la disponibilitat de les seves preses. A mesura que s'augmenta l'escala, la densitat d'aus s'utilitza com a estimador de l'hàbitat de les aus per avaluar l'exposició al risc de pèrdua d'hàbitat o pertorbació. Tot i que això és una pràctica comuna, l'eficàcia d'aquest mètode es veu compromesa si les dades observades no segueixen una distribució normal. De fet, les dades de comptatge d'animals rares vegades són normals. Per això, per al disseny d'estratègies de gestió ecològicament racionals a escala regional i local de qualsevol EIA cal considerar explícitament la variabilitat temporal i espacial de l'aparició la densitat d'aus marines (Tobin, 2004; et al Certain, 2007.).

El capítol tercer i quart d'aquesta tesi se centren en aquesta variabilitat espacial i temporal a través de l'aplicació de la Llei Exponencial de Taylor i l'anàlisi dels moviments de l'individu, respectivament.

DISCUSSIÓ

ÍNDEX DE SENSIBILITAT

El treball de Garthe i Hüppop (2004) va proposar l›avaluació quantitativa de la vulnerabilitat de les comunitats d›aus marines als parcs eòlics. Aquesta vulnerabilitat es calcula a través de l›índex de Sensibilitat de les Espècies (SSI) que se centra en la vulnerabilitat de les espècies (a nivell individual i de la població) i l'Índex de Sensibilitat als Parcs Eòlics (WSI). No obstant això, com s›ha demostrat en el capítol 1, la formulació matemàtica de l›índex principal té supòsits ocults, tant a nivell d›espècies com de la comunitat, que podrien conduir a estimacions incorrectes de la vulnerabilitat i una identificació parcial de les àrees clau.

La primera suposició és que tots els factors de risc associats a un determinat tipus de risc tenen igual importància i tenen una relació additiva. No obstant això, hi ha una diferència conceptual entre els factors inclosos en un tipus particular de risc. Prenent el risc de col·lisió com exemple, ens trobem amb dos tipus de factors de risc: els que estan directament associats al propi risc (per exemple, el percentatge de temps de vol dedicat a gran altitud) i els factors que no són importants en si mateixos, sinó com a agreujants que poden augmentar el risc preexistent (per exemple la maniobra de vol). En altres paraules, si una espècie d›au marina no vola en una altura amb risc de col·lisió, és irrellevant la maniobrabilitat de les espècies, ja que no haurà de evitar una turbina. Això significa que els factors tenen una estructura jeràrquica entre els factors de risc primaris i factors de agreujament que no poden ser tractats amb una formulació additiva. Per això es suggereix una funció alternativa que permeti estimar el risc de col·lisió i pertorbació.

La segona premissa diu que tots els tipus de risc (la sensibilitat a la col·lisió, els disturbis i la població) tenen la mateixa importància, i per tant una relació multiplicativa. La dificultat de mesurar la importància relativa del risc de col·lisió sobre el risc pertorbació justifica considerar-los com iguals, tot i que sovint es considera que el perill més important és la mortalitat per efecte de la col·lisió (Fox et al, 2006;. Christel, cert, et al. , 2012). Relacionar multiplicativament la col·lisió i el risc de pertorbació és conflictiu, perquè aquests dos tipus de riscos no depenen l›un de l›altre. Són dos aspectes independents sobre l›impacte potencial d›un OWF sobre les aus marines i, per tant, tenen diferents conseqüències ecològiques. Al vincular-los de manera multiplicativa es pot subestimar l'efecte d'un dels riscos, només perquè l›altre risc és molt baix. Per això, si els dos tipus de riscos han de ser combinats en un sol mapa és recomanable utilitzar una relació additiva. Malgrat això, considerem que per arribar a una decisió informada per a la gestió és preferible l›examen individual dels mapes de risc de col·lisió i els mapes de risc de pertorbació.

El pas final de l›índex és la integració de la vulnerabilitat de les espècies en una mesura de la vulnerabilitat d›una comunitat sencera. Per això, el tercer supòsit és que la contribució a la vulnerabilitat de la comunitat d›una determinada espècie d›aus marines és proporcional a l›abundància de registre d›aquesta espècie en un lloc determinat. Aquest enfocament volia impedir la instal·lació de parcs eòlics en zones amb altes agregacions d›aus marines. No obstant això, les espècies abundants són generalment aquells amb valors més baixos de SSI mentre que les espècies rares són aquelles amb valors més alts de SSI. Per a un lloc donat, SSI i abundància poden afectar el valor final de WSI en direccions oposades, fet que dificulta la interpretació de les variacions en l›índex. A més, l›ús del logaritme de l›abundància en comptes de l›abundància simple assumeix que un individu té més pes, en proporció, que un individu situat en un ramat de cent aus marines, i aquest supòsit no té suport ni ecològic ni ambiental. Per resoldre aquestes inconsistències, es suggereix un canvi important en la formulació basant-se en el treball de Leinster i Cobbold (2012) que presenta una mesura de diversitat basada no només en l›abundància d›espècies, sinó també en la similitud d›aquestes espècies.

Després d›incorporar els canvis en la formulació i aplicació de l›índex per a una àrea d›estudi tan gran com les costes de la Península Ibèrica, hi ha algunes conclusions i recomanacions que es poden extreure de l›experiència.

PATRONS D'AGREGACIÓ

Els mapes d'abundància permeten definir àrees amb un alt nombre d'aus (informació rellevant en l'avaluació de les ubicacions dels parcs eòlics en alta mar), però no proporcionen informació sobre els patrons espacials dinàmics i complexos de les aus marines al mar. Si bé els mapes de densitat permeten centrar-se en la detecció d'altes concentracions d'aus marines com àrees de risc potencial, l'aplicació de la llei exponencial de Taylor permet la distinció explícita entre les zones de transició i d'alimentació en el temps. En el capítol 3 hem corroborat la Llei exponencial de Taylor com a índex d'agregació en el temps i l'espai. Les àrees amb patrons d'agregació febles poden ser considerades com àrees de transició o trajectòria de vol mentre que els patrons d'agregació elevats són majoritàriament zones d'alimentació determinades per la disponibilitat puntual d'una oportunitat d'alimentar.

En vincular els patrons d›agregació amb un comportament particular es pot predir i classificar millor el risc d›establiment de parcs eòlics sobre una població d›aus marines o la comunitat. A les zones de transició, el principal risc serà la col·lisió directa i la mortalitat (Desholm i Kahlert, 2005; Hüppop et al, 2006.). En canvi, a les zones d›alimentació, la presència de les turbines es traduiria en la pèrdua d›hàbitat per a les espècies amb una forta resposta d›evitació o d›un augment de risc de col·lisió per a les espècies que experimenten un baix «efecte barrera» (Masden, Haydon, et al, 2010;. Perrow et al ., 2011). Per tant, després de la quantificació del patró d›agregació en una àrea donada, es pot avaluar el risc potencial per les regions dins d›aquesta àrea d›acord amb diferents nivells de preocupació. Les àrees amb risc de col·lisió i pèrdua d›hàbitat obtindrien el nivell de preocupació major (L3), les àrees amb risc de col·lisió tindrien el següent nivell de preocupació (L2), seguida de les àrees amb risc de pèrdua d›hàbitat (L1). Aquesta classificació es pot utilitzar després per triar la ubicació òptima d›un OWF (seleccionant zones amb preocupació mínima) o per definir un protocol de monitoratge requerit per a una ubicació OWF. A més, hi ha una evolució temporal dels patrons d›agregació que es correlaciona amb el cicle de vida de l›espècie. Si s›utilitza escenaris estacionals (per exemple, la cria, després de la cria, la migració, la hivernada) o un conjunt de mesos crítics, els escenaris temporals són més fàcils de comunicar perquè resumeixen la informació clau que pot posar en relleu l›impacte potencial d›un OWF en un moment delicat del cicle de vida de les aus marines o es pot utilitzar per recomanar mesures de mitigació durant els mesos crítics.

L›aplicació d›aquesta eina i la interpretació dels seus resultats són particularment útils per a l›avaluació d›àrees amb grans poblacions reproductores. No obstant això, seria aconsellable aplicar aquest mètode en altres escenaris per investigar possibles diferències en la interpretació dels riscos potencials associats amb els patrons d›agregació observats. També seria interessant estudiar les sortides d'aquesta eina en un corredor migratori. Finalment, tot i que els resultats del mètode són consistents amb les observacions ornitològiques i dades de comportament al Delta de l›Ebre, seria molt recomanable l›aplicació de les dades de telemetria per provar amb un conjunt de dades independents de la interpretació del comportament de llei de potència de Taylor.

No obstant això, l›estudi presentat en el capítol 3 demostra que la informació sobre les propietats de segon ordre de distribució de les espècies (és a dir, l›agregació social) proporciona informació addicional a les propietats de primer ordre (densitat) per a l›avaluació dels impactes potencials dels parcs eòlics marins.

SEGUIMENT INDIVIDUAL

Les ubicacions d'aus marines i el comportament de les aus marines són aspectes diferents. Aquest darrer és un component important que es pot extreure de les dades individuals de seguiment (Tremblay et al., 2009). Un dels objectius del capítol 4 va ser realitzar State-Space Models (SSM,. Jonsen et al, 2003) sobre dades de rastreig per satèlit com una alternativa final a les eines d›avaluació espaciotemporals. No obstant això, el rendiment dels transmissors (PTT) va ser molt limitat i heterogeni entre dispositius, probablement a causa de problemes de bateria. La mida final de la mostra va ser molt baixa i l›interval de temps entre els llocs massa llarg per aplicar els State-Space Models o per posar a prova les conclusions del capítol 3.

Malgrat els problemes tècnics i la mida limitada de la mostra, podem extreure algunes conclusions generals dels resultats de l›anàlisi espacial i temporal dels moviments de la gavina corsa Larus audouinii. Per alimentar-se, les aus marines han de superar la variabilitat en la distribució, l'abundància, la mobilitat i la previsibilitat de les seves fonts d'aliment (Bell, 1991). Per això, les espècies d'aus marines mostren un cert grau de plasticitat en el seu comportament al mar. Entre les moltes possibles respostes de comportament (per exemple, durada de viatge, distàncies de viatge, patrons de busseig), el Capítol 4 destaca la plasticitat en els patrons d'activitat temporal i l'ús de l'hàbitat. Per això cal abordar la plasticitat en l'avaluació de qualsevol parc eòlic marí.

Molts estudis han demostrat que el patró d'activitat de les aus marines no és constant al llarg del dia (per exemple Garthe et al, 2003;. Llit, 2010; Llit et al, 2012; .. Christel, Navarro, et al, 2012). Per tant, els censos d'aus marines, que han de ser efectuats amb la llum del dia i generalment seguint un horari constant, no sempre són suficients per capturar la variabilitat de les aus marines en funció dels cicles circadians. Algunes aus marines, per exemple, s'alimenten de petits peixos pelàgics, però aquest recurs només és disponible a les hores abans de l›alba, quan els bancs realitzen la seva migració vertical (Blaxter i Hunter, 1982). En aquest cas, els censos, que generalment comencen després de l'alba, no permetrien detectar una zona amb agregacions recurrents d›aus marines en aquests bancs.

De vegades se subestima la plasticitat de l'alimentació de les aus marines, tant pel que fa al seu ús de l›hàbitat com a la selecció de preses. Quan la disponibilitat de la presa és reduïda, espècies d›aus marines especialistes solen modificar la seva estratègia d›alimentació mitjançant l›extensió de la seva zona d›alimentació, el temps que passen al mar o la reducció del temps entre viatges (Lewis et al, 2001;. Schwemmer i Garthe, 2008). Per contra, les espècies generalistes poden canviar els seus hàbitats d›alimentació o canviar la seva dieta.

Appendix

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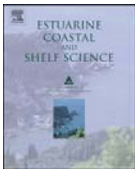

Foraging movements of Audouin's gull (*Larus audouinii*) in the Ebro Delta, NW Mediterranean: A preliminary satellite-tracking study

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ABSTRACT

A knowledge of the foraging strategies of marine predators is essential to understand the intrinsic factors controlling their distribution, abundance and their ecological function within the marine ecosystem. Here, we investigated for the first time the foraging movements and activity patterns of Audouin's gull *Larus audouinii* by using satellite-tracking data from eight breeding adults in the main colony of the species worldwide (Ebro Delta, NW Mediterranean). Tagged gulls foraged in the marine area close to the breeding colony (62% of foraging locations) and in the terrestrial area of the Ebro Delta (mainly rice fields; 38% of foraging locations). The foraging activity patterns changed significantly throughout the day; lower from dusk through the first half of the night (19-1 h; 32% of active locations) and higher during the rest of the day (1–19 h; 75.5 ± 4.3% of active locations). These results confirm the foraging plasticity of this seabird and, based on previous information about the dietary habits of this species, we hypothesize how its time-dependent activity patterns and habitat use could be associated with variations in the availability of marine food resources (e.g. diel vertical migrations of pelagic fish) and the exploitation of terrestrial resources (e.g. American crayfish *Procambarus clarkii*).

1. Introduction

An important issue in the feeding ecology of marine predators is the degree of plasticity of their foraging behavior. In general, specialist predators are constrained to forage on a specific habitat and time of day determined by a specific prey availability (Futuyma and Moreno, 1988; Krebs and Davies, 1993; Julliard et al., 2006). Under changing conditions of prey availability, specialists are able to adapt their foraging strategy by extending foraging range or time spent foraging (e.g. Oro et al., 1997; Lewis et al., 2001; Schwemmer and Garthe, 2008). By contrast, generalist predators have the ability to exploit different trophic resources and, consequently, they present higher plasticity in their foraging strategies (Krebs and Davies, 1993; Boyd et al., 2006; Julliard et al., 2006). This opportunistic behavior allows generalists to modify their foraging strategies (i.e. exploited habitat, range or temporal patterns) according, for instance, to the varying degree of competition for food. Indeed, the foraging plasticity of marine predators has allowed these organisms to benefit from anthropogenic food resources (e.g. fisheries discards, refuse dumps or introduced prey species; Tablado et al., 2010; Ramos et al., 2011; Wagner and Boersma, 2011).

Amongst marine predators, the Audouin's gull *Larus audouinii* is a good example of an opportunist species that exhibits clear plasticity in its diet habits. This Mediterranean endemic species exploits small pelagic fish (their main prey, see Oro, 1998 and references therein), but also alternative anthropogenic resources such as demersal or benthonic fish from fisheries discards or invasive freshwater crabs from terrestrial habitat (Oro et al., 1996a, 1999; Oro and Ruiz, 1997; Navarro et al., 2010). This opportunistic behavior is especially relevant in breeding populations located in areas where diverse trophic resources are highly available (e.g. Oro and Ruiz, 1997; Oro et al., 1999; Navarro et al., 2010), which is the case of the breeding population located in the Ebro Delta (Fig. 1. NW Mediterranean). This colony supports ca. 12000–13000 breeding pairs of Audouin's gull, ca. 65% of the total world population (Oro et al., 2009). The marine ecosystem of the Ebro Delta is one of the most important fishing grounds in the Mediterranean Sea, resulting in one of the largest fishing fleets in this region, which generates a high quantity of fisheries discards (Coll et al., 2008).

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Moreover, freshwater resources such as the invasive American crayfish *Procambarus clarkii* in the rice fields of the Ebro Delta are abundant and easily available (Gutierrez-Yurrita et al., 1999), providing an alternative and proficient trophic resource for the species (Oro et al., 1996b; Longoni, 2010; Navarro et al., 2010).

Although the diet habits of the Audouin's gull are well known (e.g. Oro et al., 1997; Pedrocchi et al., 2002; Sanpera et al., 2007; Navarro et al., 2010), detailed information on the foraging movements is biased toward studies based on ship surveys (e.g. Abelló and Oro, 1998; Arcos et al., 2001; Abelló et al., 2003), which are strongly biased by the influence of fishery discards and underestimate the importance of land habitat utilization. The only previous telemetric study (radio-tracking) already pointed to the apparent importance of the terrestrial habitat for the breeding population of the Ebro Delta colony (Mañosa et al., 2004).

Here, we present preliminary results of the first satellite-tracking study of Audouin's gull during the breeding season in its largest breeding colony (Ebro Delta). This paper aims to quantify the foraging range of Audouin's gull, evaluate the habitat utilization of marine and terrestrial areas and identify the temporal patterns of the foraging activity of the species. Based on previous information about the dietary habits of this species, we also hypothesize how the observed foraging movements could be attributed to the exploitation of different trophic resources in the Ebro Delta marine and terrestrial ecosystems.

2. Material and methods

2.1. Fieldwork procedures

The study was carried out at the natural reserve of Punta de la Banya in the Ebro Delta Natural Park, North Western Mediterranean Sea (Fig. 1, 40°33′N, 0°39′E). Punta de la Banya is a flat sandy peninsula of 2514 ha, partially occupied by saltworks and connected to extensive rice field areas (20,000 ha) by a 5 km-long narrow sand bar. To examine the foraging activity, we satellite-tracked 8 breeding birds (4 males and 4 females) using battery powered "Platform Transmitter Terminals" (PTTs; North Star Science and Technology, LLC) during the chick-rearing period (May) of 2006 (Table 1). We captured all birds on the nest by using a drop trap (Mills and Ryder, 1979) during late incubation to reduce the risk of desertion. Once trapped, each individual was sexed, weighed, ringed and tagged with a PTT. The attached PTTs weighed 20 g and were programmed to be active in a 6 h on/5 h off duty cycle to get information on the foraging locations during one month. The PTT was fixed to the mid-dorsal feathers of the mantle using Tesa tape (Wilson et al., 1997). With this method the PTT falls off after one month without the necessity to recapture the instrumented bird. The entire transmitter equipment represented between 3 and 4% of the Audouin's gull's body mass, so the potential effects of an additional weight on the gull's movement were minimized (e.g. Phillips et al., 2003; Passos et al., 2010).

2.2. Satellite-tracking data and statistical analyses

Data on the position of each PTT were obtained from ARGOS system (CLS, Toulouse, France) and imported to ArcView 3.2 (ESRI) using the Argos Tool extension (Potapov and Dubinin, 2005). Each position was classified according to its estimated error: Type 0 (>1000 m), Type 1 (350–1000 m), Type 2 (150–350 m), Type 3 (0–150 m), and Types A and B (without an estimated error) (ARGOS, 2006). Initial data filtering involved calculating velocities between successive satellite locations, and rejecting those for which the velocity exceeded a threshold of 50 m s^{-1}, the maximum velocity described for this species (Rosén and Hedenström, 2001). By this procedure, up to 8% of the locations were filtered; all of them from the low-quality accuracy class "B".

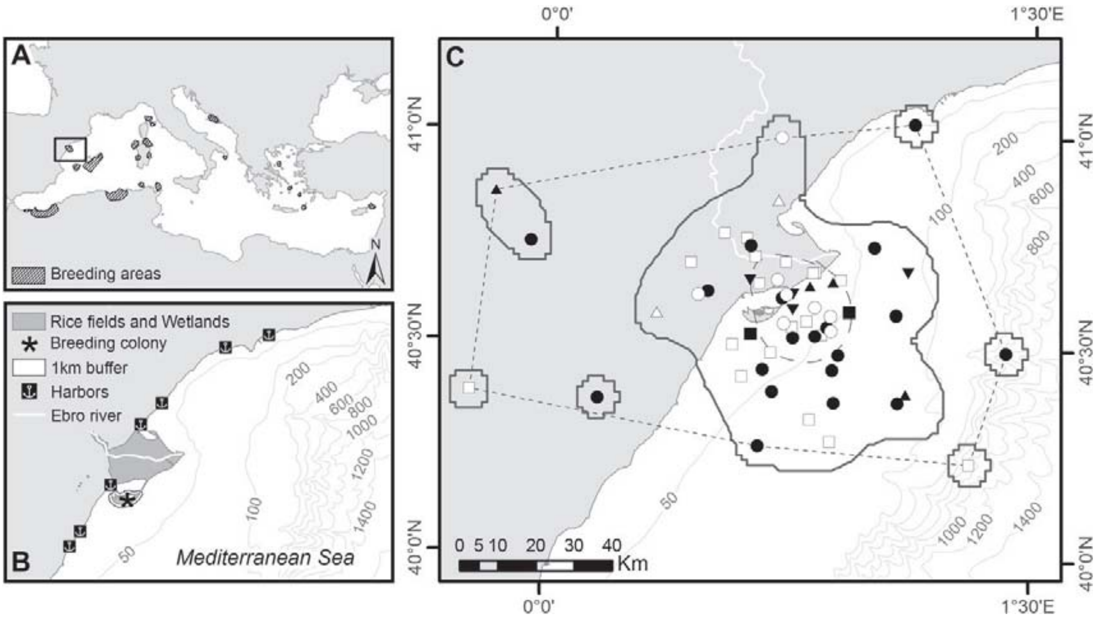

Fig. 1. (a) Breeding areas of the Mediterranean endemic Audouin's gull *Larus audouinii* and study area: Ebro Delta, NW Mediterranean. (BirdLife International, 2011) (b) Map of the Ebro Delta area indicating the Audouin's gull colony position with an asterisk and 1 km buffer area around la "Punta de la Banya" peninsula, the rice fields and wetlands shaded in dark gray and the location of the main harbors. (c) Foraging locations of 7 satellite-tracked Audouin's gulls during the breeding period of 2006. To better visualize the foraging locations' range the Minimum Convex polygon (short dashed line) is shown beside the 95% (solid line) and 50% (long dashed line) kernel polygons.

Table 1
Summary information of PTTs performance.

PTT Id	Sex	Tracking days	First location	Last location	Total locations
58978	♂	2	15/05/2006	16/05/2006	6
58979	♂	1	19/05/2006	19/05/2006	2
58980	♂	10	18/05/2006	27/05/2006	31
58981	♂	2	15/05/2006	16/05/2006	4
58982	♀	7	15/05/2006	21/05/2006	6
58983	♀	0	–	–	–
58984	♀	10	15/05/2006	24/05/2006	32
58985	♀	3	18/05/2006	20/05/2006	8
Total		13	15/05/2006	27/05/2006	89

To gain an insight into the foraging activity of the tagged Audouin's gulls we sorted the locations into three classes, according to their spatial position. PTT locations inside the "Punta de la Banya" peninsula or within the first kilometer around it were classified into the "colony locations" group. In contrast, the locations outside the colony and the first kilometer around it were "foraging locations" (we assumed that the birds were feeding to recover the body condition lost during the incubation bout). Finally, we calculated the 95% fixed-kernel estimates of the foraging area and the maximum foraging distance from the colony.

We employed logistic regression – a generalized linear model (GLM) – to test the foraging activity and habitat use. First, we tested a model with the proportion of foraging locations as the dependent variable, and we selected as the explanatory variable the "time of day" -categorized in 6-h intervals (1–7 h; 7–13 h; 13–19 h; 19-1 h)- with the 7–13 h interval as the reference level. Then we analyzed habitat use by testing the effect of the explanatory variable "time of day" on the dependent variable "terrestrial *vs.* marine proportion of foraging locations". The analyses were carried out using R software (R Development Core Team, 2008), calling the "glm" function with binomial error distribution and its default logit link function. A likelihood ratio test was used to compare the resulting model with the null model (without any variable) and to assess the significance of the explanatory variable "time of day".

3. Results

We obtained a total of 89 filtered PTT locations spanning a period of 13 consecutive days. One of the eight PTTs failed to give any location probably due to a battery failure, and the performance of the remaining PTTs was heterogeneous (see Table 1). Due to sample size limitations individual variability was not included in the analysis, but the movements of one of the tracked individuals is shown in Fig. 2 to illustrate the general pattern of the foraging movements.

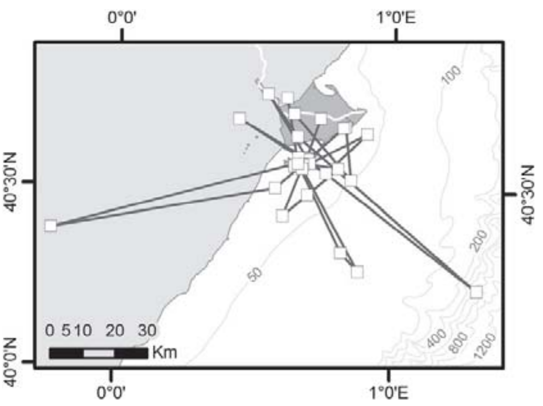

Fig. 2. Example of foraging trajectories for the individual "58980" (see Table 1 for more information).

The foraging area covered by the Audouin's gulls was 5400 km^2 (95% fixed-kernel density estimate), covering both the marine area of the Ebro Delta (ca. 3300 km^2) and the terrestrial area (ca. 2100 km^2) (Fig. 1c). The maximum foraging distance covered ranged from 20.5 to 81.7 km (mean ± sd = 51.5 ± 24.3 km) and was similar for both marine and terrestrial locations (T-Student test, $T = 1.44$, df = 56, $p = 0.15$).

The foraging activity changed significantly over the course of the day (Likelihood Ratio Test, $\chi^2 = 13.79$, df = 3, $p = 0.003$). Tagged gulls were more active at 7–13 h (78.1%), at 1–7 h (77.8% of the total locations in this period, $p = 0.65$), and 13–19 h (70.6%, $p = 0.56$), all of them significantly different from the 19-1 h interval (31.8%, $p = 0.001$), i.e., the foraging activity diminished during the first half of the night (Fig. 3b). Moreover, we found that the proportion of foraging locations in marine *vs.* terrestrial habitats changed during the day. Although the time of day was not significant as a global explanatory variable, the model indicated a significant difference between the 13–19 h interval and the reference level 7–13 h ($p = 0.04$) (Fig. 3c). Between 13 h and 19 h, Audouin's gulls foraged mainly in terrestrial (41%) rather than in marine habitat (29%); during the rest of the day, they foraged mainly in marine rather than terrestrial habitat (1–7 h: 50% marine, 28% terrestrial habitat; 7–13 h: 59% marine, 19% terrestrial habitat; 19-1 h: 23% marine, 9% terrestrial habitat) (Fig. 3a).

4. Discussion

Satellite-tracked Audouin's gulls covered a foraging area that ranges 80 km, spanning both marine and terrestrial habitats. It has been widely described previously that breeding Audouin's gulls cover large ranges when foraging. There are records of individuals foraging at 70–150 km from the breeding colony during the breeding season (Baccetti et al., 2000; Mañosa et al., 2004), and data from vessel counts suggest that individuals forage during the day and night even further offshore (Arcos and Oro, 1996; Abelló and Oro, 1998). However, the species' terrestrial foraging movements had been scarcely described (Ruiz et al., 1996; Mañosa et al., 2004).

It is well documented that Audouin's gulls forage during the night in marine habitats preying on small pelagic fish and exploiting discards provided by nocturnal fisheries (e.g. Witt et al., 1981; Mañosa et al., 2004; Arcos et al., 2008). However, our results highlight that the species' nocturnal activity is not homogeneous throughout the night (see Fig. 3). Satellite-tracked gulls were mainly located in the breeding colony during the hours before and after dusk (19-1 h). In the period after midnight to dawn (1–7 h) they increased their foraging activity, which then remained constant and high during the day. These results, coupled with the nocturnal arrival and departure times from the breeding colony described in Mañosa et al. (2004), confirm a peak of activity between midnight and dawn. Attendance to purse seiners during the night is considered a strategy that is only significant during trawling moratorium and winter periods (Arcos and Oro, 2002), neither of which were covered during our study; therefore, the individuals located at sea during the night were probably feeding on small pelagic fish. Accordingly, the nocturnal foraging habits of the Audouin's gull would still rely on the capture of small pelagic fish (Witt et al., 1981; Oro, 1998), a resource that might not be available throughout the night, but only in the hours before dawn due to the diel vertical migration of the shoals (Blaxter and Hunter, 1982; Oro, 1998).

With regard to diurnal activity, tagged birds showed a high foraging activity with an unexpected constant presence in

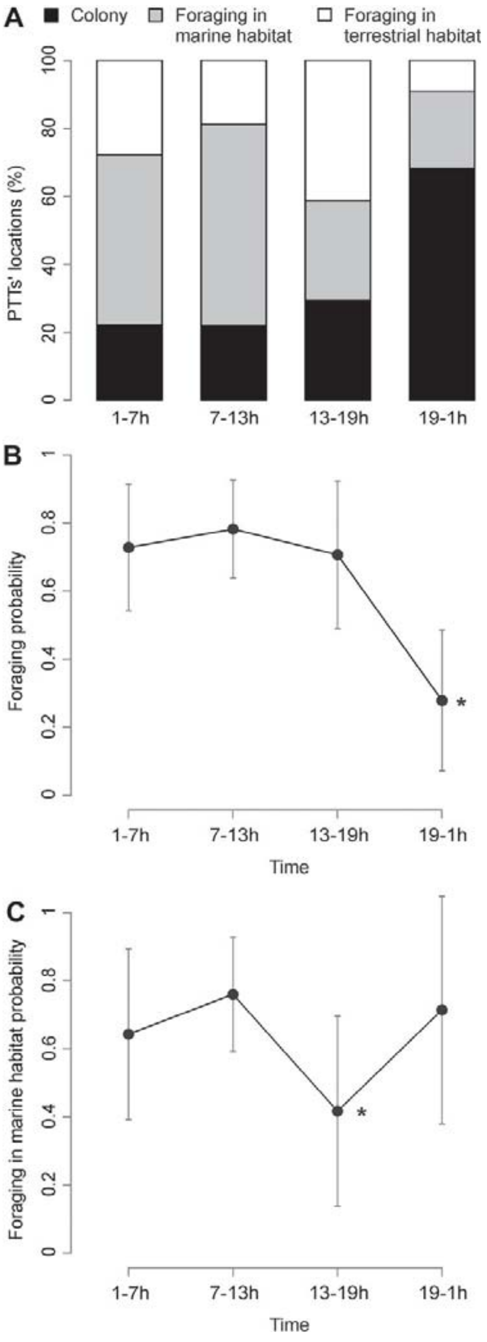

Fig. 3. (a) Activity (foraging in marine or terrestrial habitat; or located in the colony) during a 24 h cycle of 7 satellite-tracked Audouin's gulls during the breeding period in Ebro Delta colony. (b), (c) Mean and 95% confidence interval, of the foraging probability and foraging in marine habitat probability respectively, according to the GLM models. * indicates a significant difference of the time block probability compared to the reference level 7–13 h.

terrestrial habitats (generally rice fields or wetlands) in addition to the expected presence in marine habitat (Oro, 1998). The fact that all tagged individuals could be found in both habitats suggests that the use of terrestrial habitat was not due to the casual behavior of a single individual. This result supports previous studies that describe the use of the rice fields of the Ebro Delta by the Audouin's gull (Ruiz et al., 1996; Mañosa et al., 2004; Longoni, 2010), probably related to the exploitation of the exotic American crayfish (Navarro et al., 2010), which is very abundant in the rice fields of the Ebro Delta (Gutierrez-Yurrita et al., 1999). Although many studies have demonstrated that the Audouin's gull exploits trawler discards (Oro et al., 1997; Arcos, 2001; Cama, 2010), the foraging activity of our satellite-tracked individuals was higher inland than at sea in a period of time that includes the discarding peak of the trawling fleet (from 15 to 16 h; Cama, 2010). This result suggests that terrestrial foraging has become an alternative food source to trawling discards (Navarro et al., 2010), probably prompted by the interference competition for fisheries discards: namely, intraspecific competition (due to an increasing population density), and interspecific competition with the sympatric and dominant Yellow legged gull *Larus michahellis* (e.g. Arcos et al., 2001).

In conclusion, the present study shows that Audouin's gull foraged in both marine and terrestrial habitats and showed activity during both night and day. These results confirm the high foraging plasticity of Audouin's gull, a species once defined as a specialist nocturnal forager that has become an opportunist on fisheries discards and terrestrial resources. However, due the limited sample size we suggest the necessity of conducting more studies using biologging methodologies (such as PTTs or GPS) to confirm the observed patterns and to gain new insight into the foraging ecology of this endangered seabird.

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References

Abelló, P., Oro, D., 1998. Offshore distribution of seabirds in the northwestern Mediterranean in June 1995. Colonial Waterbirds 21, 422–426.

Abelló, P., Arcos, J., Gil de Sola, L., 2003. Geographical patterns of seabird attendance to a research trawler along the Iberian Mediterranean coast. Scientia Marina 67, 69–75.

Arcos, J., 2001. Foraging ecology of seabirds at sea: significance of commercial fisheries in the NW Mediterranean. PhD thesis, University of Barcelona.

Arcos, J., Oro, D., 1996. Changes in foraging range of Audouin's gulls *Larus audouinii* in relation to a trawler moratorium in the western Mediterranean. Colonial Waterbirds 1, 128–131.

Arcos, J., Oro, D., Sol, D., 2001. Competition between the yellow-legged gull *Larus cachinnans* and Audouin's gull *Larus audouinii* associated with commercial fishing vessels: the influence of season and fishing fleet. Marine Biology 139, 807–816.

Arcos, J., Oro, D., 2002. Significance of nocturnal purse seine fisheries for seabirds: a case study off the Ebro Delta (NW Mediterranean). Marine Biology 141, 277–286.

ARGOS, 2006. Argos User's Manual. ARGOS CLS, Toulouse, France. http://www.argos-system.org/manual/link. Last accessed 24 November 2011.

Arcos, J., Louzao, M., Oro, D., 2008. Fishery ecosystem impacts and management in the mediterranean: seabirds point of view. American Fisheries Society Symposium 49, 1471–1479.

Baccetti, N., Dall'Antonia, L., Magnani, A., Serra, L., 2000. Foraging routes of Audouin's Gulls Larus audouinii from two Sardinian colonies. In: Sultana, J., Yésou, P. (Eds.), Monitoring and Conservation of Birds, Mammals and Sea Turtles of the Mediterranean and Black Seas. Environment Protection Department, Malta, pp. 150–158.

BirdLife International, 2011. Distribution Maps of Birds of the World. BridLife International, Cambridge, UK.

Blaxter, J.H.S., Hunter, J.R., 1982. The biology of clupeoid fishes. Advances in Marine Biology 20, 1–223.

Boyd, I.L., Wanless, S., Camphuysen, C.J. (Eds.), 2006. Top Predators in Marine Ecosystems: Their Role in Monitoring and Management. Cambridge University Press, Cambridge, p. 378.

Cama, A., 2010. Seabird distribution patterns and ecological factors driving larid presence at the Ebro delta shelf (NW Mediterranean). PhD thesis, University of Barcelona.

Coll, M., Palomera, I., Tudela, S., Dowd, M., 2008. Food-web dynamics in the South Catalan Sea ecosystem (NW Mediterranean) for 1978–2003. Ecological Modelling 217, 95–116.

Futuyma, D.J., Moreno, G., 1988. The evolution of ecological specialization. Annual Review of Ecology and Systematics 19, 207–233.

Gutierrez-Yurrita, P., Martínez, J., Ilhéu, M., Bravo-Utrera, M., Bernardo, J., Montes, C., 1999. The status of crayfish populations in Spain and Portugal. In: Gerardi, F., Holdich, D. (Eds.), Crayfish in Europe as Alien Species: How to Make the Best of a Bad Situation? Crustacean Issues, Balkema, Rotterdam, pp. 161–192.

Julliard, R., Clavel, J., Devictor, V., Jiguet, F., Couvet, D., 2006. Spatial segregation of specialists and generalists in bird communities. Ecology Letters 9, 1237–1244.

Krebs, J.R., Davies, N.B., 1993. An Introduction to Behavioural Ecology. Wiley-Blackwell publishing, Oxford, p. 420.

Lewis, S., Sherratt, T.N., Hamer, K.C., Wanless, S., 2001. Evidence of intra-specific competition for food in a pelagic seabird. Nature 412, 816–819.

Longoni, V., 2010. Rice fields and waterbirds in the Mediterranean region and the Middle East. Waterbirds 33, 83–96.

Mañosa, S., Oro, D., Ruiz, X., 2004. Activity patterns and foraging behaviour of Audouin's gulls in the Ebro Delta, NW Mediterranean. Scientia Marina 68, 605–614.

Mills, J.A., Ryder, J.P., 1979. Trap for capturing shore and seabirds. Bird-Banding 50, 121–123.

Navarro, J., Oro, D., Bertolero, A., Genovart, M., Delgado, A., Forero, M.G., 2010. Age and sexual differences in the exploitation of two anthropogenic food resources for an opportunistic seabird. Marine Biology 157, 2453–2459.

Oro, D., 1998. *Larus audouinii* Audouin's Gull. BWP Update. Oxford University Press, Oxford, pp. 47–61.

Oro, D., Jover, L., Ruiz, X., 1996a. Influence of trawling activity on the breeding ecology of a threatened seabird, Audouin's gull *Larus audouinii*. Marine Ecology Progress Series 139, 19–29.

Oro, D., Genovart, M., Ruiz, X., Jiménez, J., García-Gans, J., 1996b. Differences in diet, population increase and breeding performance between two colonies of Audouin's gulls *Larus audouinii* during breeding seasons affected by a trawling moratorium. Journal of Avian Biology 27, 245–251.

Oro, D., Ruiz, X., 1997. Exploitation of trawler discards by breeding seabirds in the north-western Mediterranean: differences between the Ebro Delta and the Balearic Islands areas. ICES Journal of Marine Science 54, 695–707.

Oro, D., Ruiz, X., Jover, L., Pedrocchi, V., González-Solís, J., 1997. Diet and adult time budgets of Audouin's gull *Larus audouinii* in response to changes in commercial fisheries. Ibis 139, 631–637.

Oro, D., Pradel, R., Lebreton, J.D., 1999. Food availability and nest predation influence life history traits in Audouin's gull, *Larus audouinii*. Oecologia 118, 438–445.

Oro, D., Pérez-Rodríguez, A., Martínez-Vilalta, A., Bertolero, A., Vidal, F., 2009. Interference competition in a threatened seabird community: a paradox for a successful conservation. Biological Conservation 142, 1830–1835.

Passos, C., Navarro, J., Giudici, A., González-Solís, J., 2010. Effects of extra mass on the pelagic behavior of a seabird. The Auk 127, 100–107.

Pedrocchi, V., Oro, D., González-Solís, J., Ruiz, X., Jover, L., 2002. Differences in diet between the two largest breeding colonies of Audouin's gulls: the effects of fishery activities. Scientia Marina 66, 313–320.

Phillips, R., Xavier, J.C., Croxall, J.P., Burger, A.E., 2003. Effects of satellite transmitters on albatrosses and petrels. The Auk 120, 1082–1090.

Potapov, E., Dubinin, M., 2005. Argos-tools Manual. The Falcon Research Institute. National Avian Research Center, Carmarthen, UK.

Ramos, R., Ramírez, F., Carrasco, J.P., Jover, L., 2011. Insights into the spatiotemporal component of feeding ecology: an isotopic approach for conservation management sciences. Diversity and Distributions 17, 1–12.

R Development Core Team, 2008. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria, ISBN 3-900051-07-0. URL. http://www.R-project.org/.

Rosén, M., Hedenström, A., 2001. Testing predictions from flight mechanical theory: a case study of Cory's shearwater and Audouin's gull. Acta ethologica 3, 135–140.

Ruiz, X., Oro, D., Martínez-Vilalta, A., Jover, L., 1996. Feeding ecology of Audouin's gull *Larus audouinii* in the Ebro Delta. Colonial Waterbirds 19, 68–74.

Sanpera, C., Ruiz, X., Moreno, R., Jover, L., Waldron, S., 2007. Mercury and stable isotopes in feathers of Audouin's gulls as indicators of feeding habits and migratory connectivity. The Condor 109, 268–275.

Schwemmer, P., Garthe, S., 2008. Regular habitat switch as an important feeding strategy of an opportunistic seabird species at the interface between land and sea. Estuarine, Coastal and Shelf Science 77, 12–22.

Tablado, Z., Tella, J.L., Sánchez-Zapata, J.A., Hiraldo, F., 2010. The paradox of the long-term positive effects of a North American Crayfish on a European community of predators. Conservation Biology 24, 1230–1238.

Wagner, E.L., Boersma, P.D., 2011. Effects of fisheries on seabird community ecology. Reviews in Fisheries Science 19, 157–163.

Wilson, R.P., Pütz, K., Peters, G., Culik, B., Scolaro, J.A., Charrassin, J.B., Ropert-Coudert, Y., 1997. Long-term attachment of transmitting and recording devices to penguins and other seabirds. Wildlife Society Bulletin 25, 101–106.

Witt, H.H., Crespo, J., de Juana, E., Varela, J.M., 1981. Comparative feeding ecology of Audouin's gull *Larus audouinii* and the Herring gull *L. argentatus* in the Mediterranean. Ibis 123, 519–526.