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La comunidad de aves acuáticas del Mar Menor (Murcia, SE España): respuestas a procesos funcionales, gradientes estructurales e implicaciones para la gestión

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The waterbird community of Mar Menor lagoon (Murcia, SE España): responses to functional processes, structural gradients and management implications

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Tesis Doctoral

La comunidad de aves acuáticas del Mar Menor (Murcia, SE España): respuestas a procesos funcionales, gradientes estructurales e implicaciones para la gestión

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❧

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Introducción general

Los humedales

Importancia para la conservación y el bienestar humano

Los humedales son esenciales para el suministro de múltiples servicios ecosistémicos (Millenium Ecosystem Assessment, 2005). En primer lugar, son un elemento clave en la regulación del ciclo hidrológico, al participar en la descarga y recarga de los acuíferos (Maltby y Acreman, 2011), de los que con frecuencia depende estrechamente su nivel freático (Custodio, 2001). A menudo, es la cercanía del nivel freático, más que la propia presencia de agua, la que determina la tipología de humedal y la estructura de la vegetación presente (que va a caracterizar o condicionar a la comunidad faunística asociada).

Representan ecosistemas de elevado valor biológico ya que atraen a numerosos organismos de importancia para la conservación, como pueden ser taxones endémicos de macroinvertebrados, así como anfibios, peces y aves acuáticas de distribución geográfica o ecológica restringida (Gopal *et al*., 2000). Además, son enclaves de elevada producción biológica, sobre todo en áreas caracterizadas por su variabilidad como los paisajes mediterráneos, donde se mantienen activos durante las épocas desfavorables (con déficit hídrico). En estos paisajes, los humedales se definen como anomalías hídricas positivas (González-Bernáldez y Montes, 1989) cuya permanencia durante los periodos desfavorables aporta un elemento paisajístico de tipo “oasis”, manteniendo una productividad biológica elevada con respecto al territorio circundante. Esto apoya el mantenimiento de redes de humedales como componentes esenciales del paisaje para la mantener la biodiversidad.

Sin embargo, y a pesar de los valores comentados y otros servicios evidentes de regulación y abastecimiento (filtros verdes, producción de sal, riego, etc.), desde la perspectiva de la cultura humana, los humedales siguen siendo un tipo de paisaje infravalorado, al menos en su percepción “estructural” directa por la sociedad, frente a otros como bosques o lagos (Nassauer, 2004). Esto refuerza la necesidad de manifestar su importancia para la conservación, ya que siendo medios naturales aprovechados por el ser humano en múltiples aspectos, no son suficientemente reconocidos por éste como paisajes valiosos y frágiles, por tanto, muy susceptibles de ser degradados por un uso o gestión inadecuados.

Salinas de Marchamalo. Fotografía: Pablo Farinós Celdrán

Factores que condicionan su grado de amenaza

Pese a su papel clave para la biodiversidad y las sociedades humanas, los humedales se encuentran entre los tipos de ecosistemas más amenazados del planeta (Millenium Ecosystem Assessment, 2005), debido a la confluencia de varios factores:

- Al ser cuerpos de agua, el uso de sus cubetas y cuencas vertientes se remonta muchos siglos en la historia, encontrándose entre los ecosistemas que más tiempo llevan soportando el aprovechamiento humano, con las presiones y cambios que eso suele conllevar (sobreexplotación de recursos –vivos o inertes-, cambios físicos y morfológicos, cambios en la naturaleza fisicoquímica del agua, etc.). De la misma manera, especialmente en el ámbito mediterráneo, donde la gestión tradicional ha contribuido a la diversificación de servicios ecosistémicos, el abandono de usos puede traer consigo efectos tan negativos como su intensificación (Zorrilla-Miras *et al*., 2014).

- Por otro lado, muchos humedales como las lagunas costeras, salinas, criptohumedales litorales, representan hábitats de transición o ecotonos entre grandes sistemas ecológicos, como son el medio terrestre-continental y el medio marino-oceánico (Naiman y Decamps, 1990). Esto hace que sean ecosistemas muy dinámicos y cambiantes, con una alta capacidad de autorregulación y autodepuración, que en algunos casos se manifiesta como amortiguadora o “tamponadora” ante determinados procesos que operan entre esos dos grandes compartimentos (Bugenyi, 2001). Todas estas son características que los definen como sistemas de alerta, al manifestar de forma integrada y relativamente rápida los efectos derivados de cualquier proceso físico-químico en diferentes escalas espaciales y temporales (Esteve *et al*., 2008). Ejemplos de ese elevado dinamismo se pueden encontrar en el conjunto de respuestas físicas, químicas y biológicas que un humedal puede mostrar frente a vertidos y contaminantes de muy diferente naturaleza (Dordio *et al*., 2008). Otro ejemplo, desde una perspectiva espacial más amplia, es el frágil equilibrio mantenido entre los acuíferos, los humedales litorales y el medio marino, que puede verse fácilmente alterado ante determinados modelos de gestión del agua (Rey *et al*., 2013), provocando drásticos cambios en los gradientes internos, como la salinidad, profundidad, etc.

❖ Además, gran parte de las especies que conforman sus comunidades biológicas manifiestan un alto grado de especialización (de ahí, en muchos casos, su importancia para la conservación). Esto las hace muy sensibles a los cambios ambientales, e incapaces de adaptarse o de hacer frente a la competencia de especies más generalistas beneficiadas por dichos cambios, llegando a producirse sustituciones de especies o cambios demográficos que pueden transmitirse a lo largo de la red trófica. Por tanto, se podría decir que al ser ecosistemas tan dinámicos, sus comunidades biológicas son estructural y funcionalmente muy sensibles al cambio ambiental. En concreto, las aves acuáticas, como grupo faunístico dependiente de los humedales, es una de las taxocenosis que más evidencia esa “sensibilidad” ambiental, que puede manifestarse en una disminución de los taxones especializados o en un incremento de los generalistas (Amat y Green, 2010). Su naturaleza migratoria les confiere además el carácter de patrimonio transnacional. Seguramente por ello, son uno de los grupos faunísticos que la sociedad prioriza en cuanto a su seguimiento, estudio, gestión y conservación.

Flamencos *Phoenicopterus roseus* y Tarros blancos *Tadorna tadorna* alimentándose en una charca del Parque Regional Salinas y Arenales de San Pedro del Pinatar. Fotografía: Pablo Farinós Celdrán

Marcos de conservación y gestión de humedales

La importancia que los humedales tienen para la preservación de la biodiversidad mundial y el control de procesos biogeoquímicos, en base a las características descritas y en relación con los factores de presión identificados, ha llevado al hombre a reconocer la necesidad de preservar este tipo de ecosistemas. En la actualidad, la protección y conservación de humedales no sólo se orienta a defender sus valores frente a los factores de presión, sino a promover su uso racional. Esto incluye su utilización como espacios de seguimiento e investigación, mediante el estudio de organismos bioindicadores, la monitorización de sus características fisicoquímicas y otras variables (Ramsar Convention Secretariat, 2013) ya que, como se ha apuntado anteriormente, son enclaves que evidencian de forma rápida los efectos de cualquier proceso, impacto o alteración. Además, en un escenario de cambio climático global, su elevado dinamismo y capacidad de autorregulación, de cambiar condiciones y de acelerar tasas de cambio, va a suponer un aspecto crucial frente a los efectos a medio y largo plazo de dicho proceso (Lloret *et al*., 2008). En este contexto, es conveniente pensar en un sistema global de protección de este tipo de ecosistemas (Amezaga *et al*., 2002; Ramsar Convention Secretariat, 2013), en especial cuando se refiere a un componente espacial y temporalmente tan dinámico como las aves acuáticas. Resulta fundamental armonizar los objetivos de los diferentes marcos legales de protección internacional de los humedales, por ejemplo, integrando la monitorización de las aves acuáticas en la evaluación del estado ecológico de los humedales en el contexto de la Directiva Marco del Agua (WFD, European Comission, 2000) dado que son el taxón prioritario para la conservación en otras normativas internacionales como la Directiva

Aves (Birds Directive 2009/147/CE) o el convenio Ramsar (Ramsar Convention Secretariat, 2013) (Uttley, 2010).

Por tanto, parece claro que la gestión de los humedales se debe basar en un enfoque de conservación integrando diferentes escalas espaciales y niveles de organización biológica y ecológica. Un enfoque que aúne tanto la conservación del propio humedal, sus gradientes internos y sus comunidades biológicas (y los servicios que ofrecen), como la conservación del paisaje circundante y la planificación y gestión sostenible de las actividades humanas que se dan en el mismo, y finalmente la gestión integral de redes de humedales manteniendo su conectividad ecológica. En este sentido, la dispersión espacial de los humedales en el paisaje favorece su utilización por taxocenosis de alta movilidad como las aves acuáticas, que acoplan sus ciclos vitales a la dinámica espacial y temporal de estos hábitats y contribuyen a su conectividad biológica (Amezaga *et al*., 2003). Se espera por lo tanto que cualquier intervención que suponga un cambio en la manifestación espacial del agua en el paisaje (irrigación, regulación, sobreexplotación…), traiga consigo una respuesta protagonizada por estas taxocenosis. La monitorización de estas respuestas puede suponer una herramienta incomparable a la hora de evaluar los efectos de la gestión que el ser humano hace de los recursos y servicios que ofrecen los humedales y otros ecosistemas asociados al agua.

Las aves acuáticas

Las aves acuáticas son una de las taxocensosis de mayor interés para la conservación internacional (Birds Directive 2009/147/CE), siendo utilizadas como criterio principal para designar humedales de importancia internacional (Ramsar Convention Secretariat, 2013). No obstante, una vez que los humedales reciben una categoría de protección basada en esas normas o acuerdos (Humedal Ramsar de Importancia Internacional, Zona de Especial Protección para las Aves) los datos obtenidos mediante el seguimiento de las aves acuáticas suelen ser evaluados de forma aislada (muchas veces en términos puramente cuantitativos) y presentados como simples indicadores de “éxito” en cuanto al tipo de gestión que han recibido las aves y sus hábitats. Es mucho menos frecuente que los datos obtenidos con ese seguimiento se analicen de forma integral, relacionándolos con descriptores de otros componentes del sistema (otras taxocenosis o factores abióticos) (Adamus, 1996; Amat y Green, 2010). Sin embargo, en muchos casos se ha demostrado que las aves pueden relacionarse muy directamente con el estatus trófico, la abundancia de vegetación acuática o la abundancia de peces (Amat y Green, 2010). De hecho, las aves acuáticas pueden ser determinantes directos de la diversidad de otros organismos, actuar como vectores de dispersión de animales o plantas, o participar en el balance de nutrientes, así como otra serie de servicios ecosistémicos que incluso repercuten positivamente en las sociedades humanas (Green y Elmberg, 2014).

Garceta común *Egretta garzetta* cazando en la ribera de la laguna del Mar Menor. Fotografía: Pablo Farinós Celdrán

El papel bioindicador de las aves acuáticas ha sido ampliamente discutido en muchos estudios (Green y Figuerola, 2003; Gregory *et al*., 2003; Gregory *et al*., 2005) y su utilidad como herramienta de monitorización y de “alarma” se relaciona con una serie de potencialidades (respuestas en diferentes escalas espaciales, consumidores apicales, etc.) pero también con limitaciones (elevada movilidad, diferencias entre índices de diversidad basados en aves y en otros organismos) (Amat y Green, 2010). Lo que parece evidente es que las aves acuáticas responden a cambios en las condiciones del medio, debidos a la confluencia de varios factores de estrés y presión, constituyendo "señales de alarma" frente a cambios indeseables, respuestas que pueden ser monitorizadas a diferentes escalas temporales y espaciales y en diferentes niveles de organización (Kushlan, 1993; Amat y Green, 2010).

Zampullín cuellinegro *Podiceps nigricollis* nadando en aguas someras de la laguna del Mar Menor.
Fotografía: Francisca Carreño Fructuoso

En los ambientes litorales semiconfinados, como las lagunas costeras (Newton *et al*. 2014), el papel bioindicador de las aves acuáticas ha sido con frecuencia ignorado, renunciando así a su uso como herramienta de evaluación y monitorización. Esto sucede pese a que constituyen una de las taxocenosis prioritarias a la hora de establecer planes de gestión y conservación en estos ambientes, cuyo papel bioindicador ha sido por otra parte reconocido y analizado específicamente (Stolen *et al*., 2004). En lo que respecta a la relación de las aves acuáticas con otras taxocenosis y el ecosistema en su conjunto, resulta evidente la diversidad y utilidad práctica de los resultados que se pueden obtener acoplando diferentes escalas de estudio y niveles de organización biológica (Robledano y Farinós, 2011). Por lo tanto, es necesario desentrañar tanto la relación de las aves con la estructura de los hábitats lagunares, como su relación con las biocenosis presentes, identificando qué parte de las respuestas es debida a condiciones y cambios locales, y cuál es debida a factores comunes y extrapolables a otros contextos biogeográficos (Robledano y Farinós, 2011).

A este potencial bioindicador, hay que añadir que las aves acuáticas son una de las taxocenosis que más tiempo llevan siendo estudiadas en relación a su distribución y ecología (Peakall y Boyd, 1987), generando multitud de registros y series temporales largas de datos, gracias en parte al gran número de estudiosos no profesionales que contribuyen a monitorizar las poblaciones y recoger esos datos. Es de destacar por tanto el papel que la "ciencia ciudadana" ha tenido en este sentido (Kushlan, 1993). Este hecho supone un argumento incontestable cuando se discute la idoneidad de las aves acuáticas como herramienta para el seguimiento y la monitorización del estado de conservación de los humedales. A esto se añade, además, que es un grupo faunístico percibido, por lo general, de forma positiva por la sociedad (Kushlan, 1993).

Gaviota picofina *Larus genei* sobre posadero en la ribera de la laguna en el PR Salinas y Arenales de San Pedro del Pinatar. Fotografía: Francisca Carreño Fructuoso

Caso de estudio: Laguna costera del Mar Menor

Una vez descritas las características principales que identifican a los humedales como ecosistemas fundamentales para preservar la biodiversidad, y habiendo puesto de manifiesto la necesidad de monitorizar y evaluar los efectos que las actividades humanas (y otros procesos naturales) tienen sobre éstos y sus comunidades biológicas asociadas, podemos focalizar el presente trabajo en un caso de estudio concreto: el Mar Menor, la mayor laguna costera del Mediterráneo occidental (Figura 1).

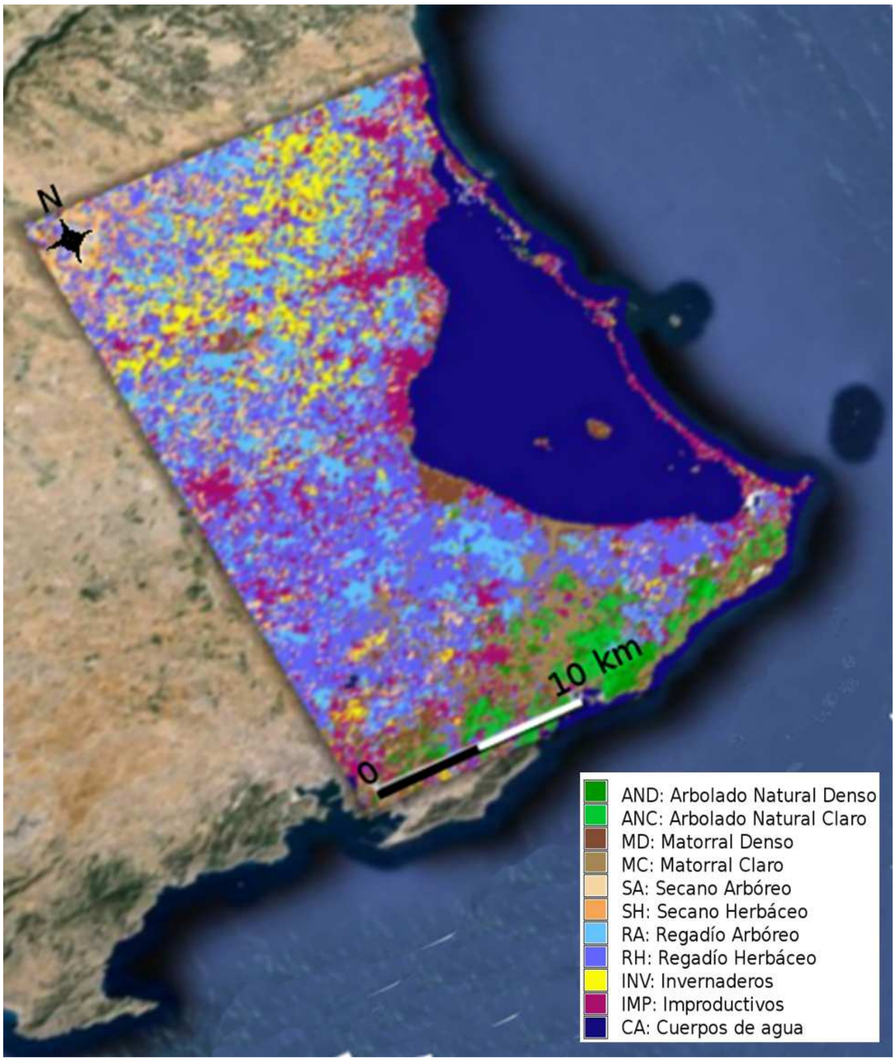


Figura 1. Imagen de satélite de la laguna de la laguna del Mar Menor mostrando los usos del suelo predominantes en su cuenca vertiente. Autora: Mª Francisca Carreño Fructuoso

Como ambiente costero de transición entre el medio marino y terrestre, en el Mar Menor se pueden encontrar diferentes gradientes ambientales que, combinados, definen un espacio multidimensional con un gran número de factores determinantes de las condiciones, en muchos casos extremas, del medio, así como una elevada variedad de efectos y respuestas en distintas escalas (espaciales y temporales). Estos gradientes se generan tanto por sus características naturales (grado de confinamiento, biocenosis dominante, salinidad, etc.) como por la confluencia de ciertas actividades de origen antrópico, que podemos aglutinar en grandes “factores de presión”.

Factores antrópicos de presión

En la tabla 1 se presentan, agrupados según la naturaleza de la actividad humana descrita, los principales factores de presión antrópicos sobre la laguna y una breve explicación de los principales efectos que produce cada uno. A su vez, se presentan los principales estudios científicos realizados en cada uno de esos campos durante las dos últimas décadas aproximadamente. Este conjunto de publicaciones supone para el presente trabajo, un marco de referencia, tanto de la relación de la avifauna con factores ambientales antrópicos, como de las relaciones entre otros componentes bióticos y/o abióticos y determinados factores de influencia antrópica que, en mayor o menor medida, pueden afectar directa o indirectamente a las aves acuáticas. La revisión no pretende ser exhaustiva, por lo que se remite a los interesados a otras publicaciones, como la de Martínez *et al.* (2003). En ella se presenta una amplia recopilación de trabajos que han estudiado (de forma preliminar en algunos casos) diferentes compartimentos del ecosistema lagunar a lo largo del siglo XX, lo que amplía significativamente este marco de referencia.

Factor	**Descripción**	**Bibliografía de referencia**
Influencia agrícola	Descarga de efluentes de origen agrícola y alteraciones tróficas en la laguna (eutrofización) Alteraciones en la estructura y/o funcionalidad de los hábitats terrestres ligados a la laguna (pérdida de ambientes esteparios y aumento de formaciones hidrófilas), así como pérdida general de diversidad paisajística, debido a los cambios de uso Alteración del equilibrio hidrológico (aumento del nivel freático)	Alvarez-Rogel *et al.* 2006; 2007; Campillo *et al.* 2013; Carreño *et al.* 2008; Conesa y Jiménez-Cárceles, 2007. Esteve *et al.* 2008; Farinós *et al.* 2009; García-Pintado *et al.* 2007: García-Sánchez *et al.* 2012; Hernández y Robledano, 1997; León *et al.* 2013; Lloret *et al.* 2005; 2008; Lloret y Marín, 2009; 2011; Martínez *et al.* 2005; 2007; 2013; Moreno *et al.* 2013; Pardo *et al.* 2008; Pérez-Ruzafa *et al.* 2000; 2002; 2004; 2005; 2007a; 2007b; 2012; Robledano *et al.* 2008; 2010; Robledano-Aymerich y Farinós-Celdrán, 2010; Rosique, 2000; Ruiz y Velasco, 2010; Salas *et al.* 2005; 2006; Velasco *et al.* 2004
Influencia urbana	Descarga de efluentes de origen urbano/industrial Fragmentación y/o destrucción de hábitats por urbanización y construcción de infraestructuras	Alvarez-Rogel *et al.* 2006; Conesa y Jiménez-Cárceles, 2007. Esteve *et al.* 2008; García-Pintado *et al.* 2007: García-Sánchez *et al.* 2012; Hernández y Robledano, 1997; León *et al.* 2013; Lloret *et al.* 2008: Lloret y Marín, 2009; 2011; Martínez *et al.* 2005; 2007; 2013; Pérez-Ruzafa *et al.* 1991; Pérez-Ruzafa

Factor	Descripción	Bibliografía de referencia
	antrópicas Alteraciones del equilibrio hidrológico Generación de residuos sólidos y vertido en condiciones no controladas Estrés por uso antrópico del humedal y su ribera	*et al.* 2006; 2007a; 2007b; 2012; Rosique, 2000; Salas *et al.* 2005; 2006.
Influencia minera	Cambios en la composición de los suelos de la ribera y en el sedimento lagunar por la lixiviación y transporte de metales pesados en la ribera sur Bioacumulación y biomagnificación en cadenas tróficas	María-Cervantes *et al.* 2009; Marín-Guirao *et al.* 2005a; 2005b; 2008; Martínez *et al.* 2003; Navarro *et al.* 2010; Rosique, 2000; Salas *et al.* 2005; 2006;

Tabla 1. Principales factores de presión de origen antrópico sobre la laguna del Mar Menor y sus humedales asociados. También se citan los principales estudios científicos realizados en cada campo.

Comunidad de aves acuáticas estudiada

A lo largo del presente trabajo, los diferentes capítulos están enfocados a estudiar la los gradientes espaciales y temporales que confluyen en la laguna y cómo afectan a la distribución, preferencias ecológicas, composición y estructura de la comunidad de aves acuáticas. No obstante, y dada la riqueza y heterogeneidad de la comunidad de aves acuáticas estudiada, se presentan a continuación todas las especies que han sido objeto de estudio a lo largo de este trabajo (tabla 2). Se indican las categorías de amenaza/protección que tiene cada especie en diferentes evaluaciones y normas legales de diferente ámbito, como representación general de su interés para la conservación. También se indica el(los) capítulo(s) de la presente memoria en donde se trata, de forma más o menos directa y con mayor o menor profundidad (en función de la escala de estudio y de los gradientes analizados) cada una de esas especies.

Nombre científico	Nombre común	Directiva Aves 47/2009/CE [1]	BirdLife International 2004 [2]	Libro Rojo de las Aves de España [3]	R.D. 139/2011 [4]	CREA Ley 7/95 [5]	Cap.
Haematopus ostralegus	Ostrero		NON SPEC	NT	R.P.E.		Cap II
Himantopus himantopus	Cigüeñuela	Anexo I	NON SPEC	No eval.	R.P.E.		Cap II
Recuvirostra avosetta	Avoceta	Anexo I	NON SPEC	LC	R.P.E.	VU	Cap II
Burhinus oedicnemus	Alcaraván	Anexo I	SPEC 3	NT	R.P.E.		Cap II
Charadrius dubius	Chorlitejo Chico		NON SPEC	No eval.	R.P.E.		Cap II
Charadrius hiaticula	Chorlitejo Grande		NON SPEC	No eval.	R.P.E.		Cap II
Charadrius alexandrinus	Chorlitejo Patinegro	Anexo I	SPEC 3	VU	R.P.E.	IE	Cap II
Pluvialis squaratola	Chorlito Gris		NON SPEC	No eval.	R.P.E.		Cap II
Vanellus vanellus	Avefría	Anexo II B (España)	SPEC 2	LC			Cap II
Calidris alba	Correlimos Tridáctilo		NON SPEC	No eval.	R.P.E.		Cap II
Calidris canutus	Correlimos Gordo		SPEC 3	No eval.	R.P.E.		Cap II
Calidris minuta	Correlimos Menudo		NON SPEC	No eval.	R.P.E.		Cap II
Calidris ferruginea	Correlimos Zarapitín		NON SPEC	No eval.	R.P.E.		Cap II
Calidris alpina	Correlimos Común		SPEC 3	No eval.	R.P.E.		Cap II
Philomachus pugnax	Combatiente	Anexo I	SPEC 2	No eval.	R.P.E.		Cap II
Gallinago gallinago	Agachadiza común	Anexo II A, Anexo III B	SPEC 3	EN	R.P.E.		Cap II
Limosa limosa	Aguja colinegra	Anexo I	SPEC 2	VU	R.P.E.		Cap II
Limosa lapponica	Aguja Colipinta		NON SPEC	No eval.	R.P.E.		Cap II
Numenius arquata	Zarapito Real		SPEC 2	EN	R.P.E.		Cap II
Tringa erythropus	Archibebe Oscuro		SPEC 3	No eval.	R.P.E.		Cap II
Tringa totanus	Archibebe Común		SPEC 2	VU	R.P.E.		Cap II
Tringa stagnatillis	Archibebe Fino		NON SPEC	No eval.	R.P.E.		Cap II
Tringa nebularia	Archibebe Claro	Anexo II B	NON SPEC	No eval.	R.P.E.		Cap II
Actitis hypoleucos	Andarríos Chico		SPEC 3	No eval.	R.P.E.		Cap II
Arenaria interpres	Vuelvepiedras		NON SPEC	No eval.	R.P.E.		Cap II
Fulica atra	Focha Común	Anexo II A, Anexo III B	NON SPEC	No eval.	R.P.E.		Cap I y IV
Gallinula chloropus	Gallineta Común		NON SPEC	No eval.	R.P.E.		Cap II

Nombre científico	**Nombre común**	**Directiva Aves 47/2009/CE** [1]	**BirdLife International 2004** [2]	**Libro Rojo de las Aves de España** [3]	**R.D. 139/2011** [4]	**CREA Ley 7/95** [5]	**Cap.**
Larus michaellis	Gaviota Patiamarilla	Anexo II B (España)	NON SPEC	No eval.			Cap II, III y IV
Larus ridibundus	Gaviota Reidora	Anexo II B (España)	NON SPEC	No eval.			Cap II, III y IV
Larus genei	Gaviota Picofina	Anexo I	SPEC 3	VU	R.P.E.		Cap II, III y IV
Larus melanocephalus	Gaviota Cabecinegra	Anexo I	NON SPEC	No eval.	R.P.E.		Cap II
Larus audouini	Gaviota de Audouin	Anexo I	SPEC 1	VU	VU	VU	Cap II, III y IV
Larus fuscus	Gaviota Sombría		NON SPEC	LC	R.P.E.		Cap II
Gelochelidon nilotica	Pagaza piconegra	Anexo I	SPEC 3	VU	R.P.E.		Cap II
Sterna caspia	Pagaza Piquirroja	Anexo I	SPEC 3	No eval.	R.P.E.		Cap II
Sterna sandvicensis	Charrán Patinegro	Anexo I	SPEC 2	NT	R.P.E.		Cap II, III y IV
Sterna hirundo	Charrán común	Anexo I	NON SPEC	NT	R.P.E.	IE	Cap II, III y IV
Sterna albifrons	Charrancito	Anexo I	SPEC 3	NT	R.P.E.	VU	Cap II y IV
Podiceps cristatus	Somormujo Lavanco		NON SPEC	No eval.	R.P.E.		Cap I, III y IV
Podiceps nigricollis	Zampullín Cuellinegro		NON SPEC	NT	R.P.E.		Cap I, III y IV
Phalacrocorax carbo	Cormorán Grande		NON SPEC	No eval.	R.P.E.		Cap I, II, III y IV
Tadorna tadorna	Tarro Blanco		NON SPEC	NT	R.P.E.	IE	Cap II y IV
Mergus serrator	Serreta Mediana		NON SPEC	No eval.	R.P.E.		Cap I, III y IV
Anas crecca	Cerceta Común	Anexo II A, Anexo III B	NON SPEC	VU	R.P.E.		Cap II
Anas acuta	Anade Rabudo	Anexo II A, Anexo III B	SPEC 3	VU	R.P.E.		Cap II
Anas platyrhynchos	Ánade Azulón o Real	Anexo II A, Anexo III A	NON SPEC	No eval.	R.P.E.		Cap II y IV
Egretta garzetta	Garceta Común	Anexo I	NON SPEC	No eval.	R.P.E.		Cap II, III y IV
Ardea alba	Garceta Grande	Anexo I	NON SPEC	No eval.	R.P.E.		Cap II y IV

Nombre científico	**Nombre común**	**Directiva Aves 47/2009/CE** [1]	**BirdLife International 2004** [2]	**Libro Rojo de las Aves de España** [3]	**R.D. 139/2011** [4]	**CREA Ley 7/95** [5]	**Cap.**
Ardea cinerea	Garza Real		NON SPEC	No eval.	R.P.E.	IE	Cap II y IV

Tabla 2. Especies de aves acuáticas estudiadas a lo largo de este trabajo. Se indican las categorías de amenaza en evaluaciones o normas legales de diferente ámbito. Categorías SPEC: SPEC 1 = Especies de preocupación para la conservación global; SPEC 2 = Concentradas en Europa y con un estado de conservación desfavorable; SPEC 3 = No concentradas en Europa pero con un estado de conservación desfavorable; NON SPEC = No concentradas en Europa y con un estado de conservación favorable. EN = En peligro; VU = Vulnerable; NT = Casi amenazado; LC = Preocupación menor. R.P.E. = Régimen de Protección Especial. IE = Interés especial

[1] Directiva 2009/147/CE del Parlamento Europeo y del Consejo de 30 de noviembre de 2009 relativa a la conservación de aves silvestres. Anexo I: especies de aves objeto de medidas de conservación; Anexo II: especies cazables en todo el territorio de la Unión (A) o en el estado miembro que se especifica (B); Anexo III: especies que podrán ser objeto de venta en todo el territorio de la Unión (A) u objeto de venta en los estados miembros previo informe de la Comisión (B)

[2] BirdLife International. 2004. Birds in the European Union: a status assessment. BirdLife International. Wageningen, The Netherlands.

[3] Madroño, A., González, C. y Atienza, J.C. (Eds.) 2004. Libro Rojo de las Aves de España. Dirección General para la Biodiversidad-SEO/BirdLife. Madrid.

[4] Real Decreto 139/2011, de 4 de febrero, para el desarrollo del Listado de Especies Silvestres en Régimen de Protección Especial y del Catálogo Español de Especies Amenazadas y del Catálogo Español de Especies Amenazadas (CEEA)

[5] Catálogo Regional de Especies Amenazadas (CREA). Ley 7/1995, de 21 de abril, de la fauna silvestre, caza y pesca fluvial (Comunidad Autónoma de la Región de Murcia)

Objetivos y estructura de la presente memoria

El objetivo general de la presente memoria fue analizar la respuesta estructural y funcional de la comunidad de aves acuáticas a diferentes gradientes naturales y antrópicos en el complejo laguna del Mar Menor-humedales asociados. Se analizó, tanto la respuesta global de la comunidad (estudiada a través de índices biológicos), como el papel indicador de ciertas especies "focales" frente a dichos gradientes. Este objetivo principal fue desglosado en cuatro objetivos específicos, atendiendo a las diferentes escalas espaciales y temporales analizadas.

- ✓ Analizar a largo plazo la influencia de ciertos factores ambientales relacionados con la productividad del sistema (carga de nutrientes, producción pesquera) sobre la evolución poblacional de determinadas especies representativas, seleccionadas para testar su posible papel bioindicador de estados de alteración trófica en las aguas de la laguna.

- ✓ Estudiar la distribución y estructura de la comunidad de aves acuáticas que domina la masa lagunar, con respecto a características funcionales internas y estructurales de la laguna, para detectar los principales gradientes biológicos (diversidad, abundancia) y su relación con dichas características.

- ✓ Describir los patrones de cambio de la comunidad de aves acuáticas a través del análisis de determinados índices y parámetros biológicos en dos escalas espaciales: humedal individual (escala local) y complejo lagunar (escala de paisaje), identificando fases de cambio ambiental y con una aplicación directa sobre la conservación integral del sistema.

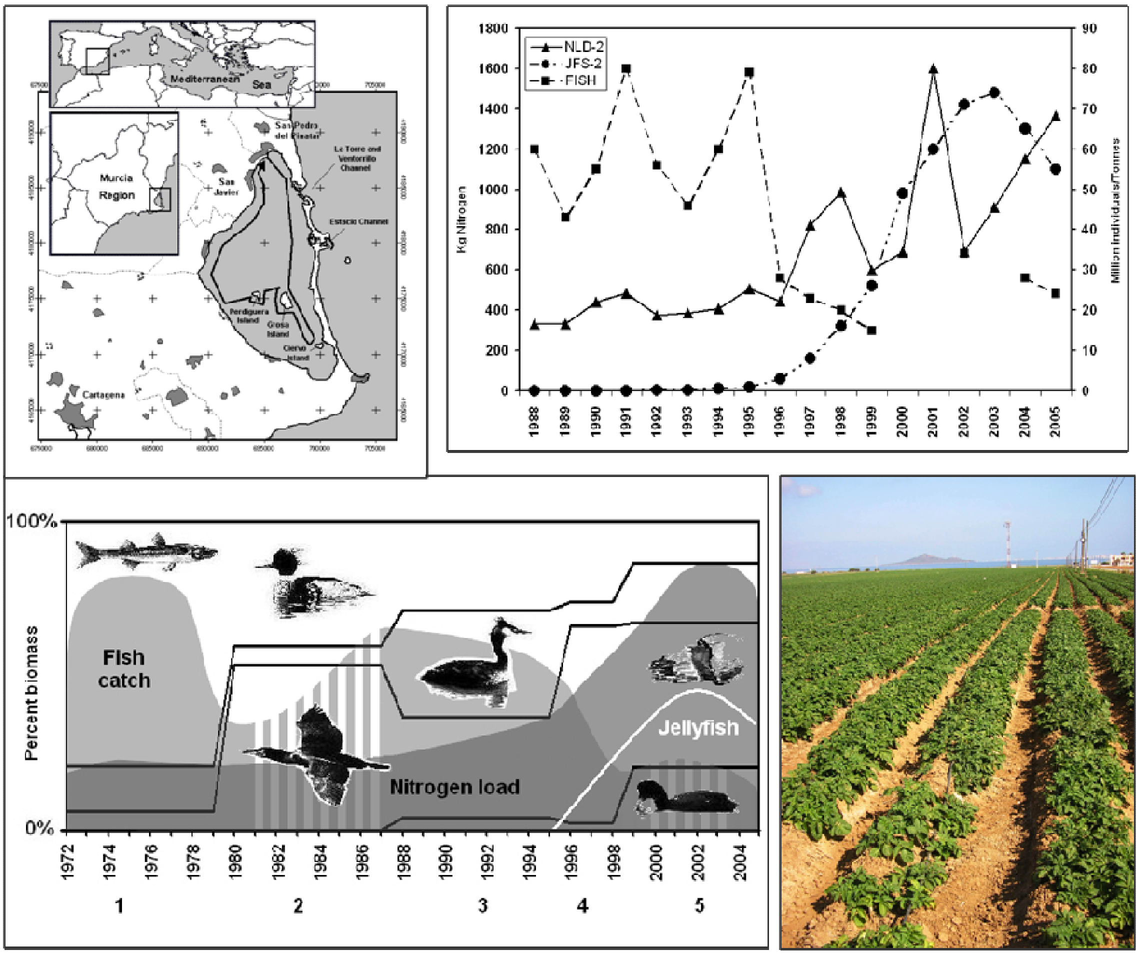
- ✓ Modelizar, a diferentes escalas, la respuesta espacial y estructural de la comunidad de aves acuáticas y la respuesta específica de determinadas especies seleccionadas a los gradientes internos de la laguna y a la estructura del paisaje que conforman sus hábitats terrestres adyacentes.

- ✓ Proponer un esquema de manejo y gestión del complejo lagunar, en el que se consiga armonizar el mantenimiento de la funcionalidad del sistema (por ejemplo, manteniendo la capacidad de depuración) con la recuperación de la singularidad de sus comunidades biológicas originales

Los cuatro primeros objetivos específicos se corresponden con los cuatro capítulos en los que se divide la presente memoria. El quinto objetivo constituye una propuesta adicional que se relaciona con los anteriores y que se concreta en las conclusiones. A continuación se presenta un resumen gráfico y en texto de dichos capítulos.

Capítulo 1. Determinantes de los cambios en la invernada de las aves acuáticas en una laguna costera Mediterránea afectada por la eutrofización.

Resumen



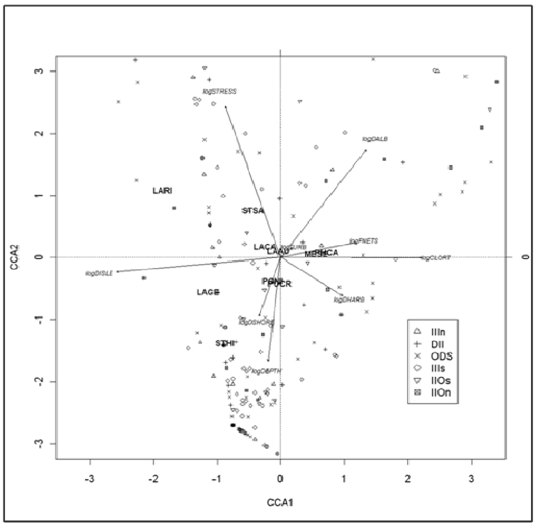
Las lagunas del sur del Mediterráneo están entre los sistemas más vulnerables en relación a las actividades humanas (agricultura, turismo, urbanismo, etc.) que producen eutrofización. Se analiza la relación de las aves acuáticas con variables ambientales medidas o calculadas localmente (concentración de nutrientes, producción de peces, proliferaciones de medusas) relacionados con este proceso, en la laguna del Mar Menor (Murcia , SE España), y se discute el valor potencial de las aves como indicadores de el estado trófico del humedal. Se utilizaron Modelos Lineales

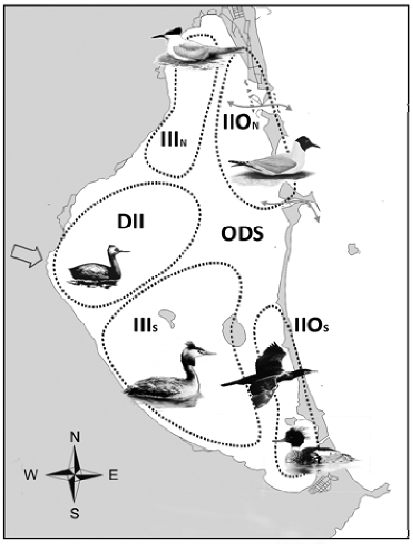
Generalizados (GLM) para relacionar la biomasa de las cinco especies de aves acuáticas más abundantes y representativas con estas variables, considerando la posible influencia de factores externos que afecten a sus poblaciones a una escala biogeográfica más amplia. Sólo se encontró un efecto significativo de estos factores para el Cormorán grande, que sesgó la respuesta positiva del gremio de los piscívoros a la carga de nutrientes (NLD). La Serreta mediana se mostró relativamente insensible al enriquecimiento de nutrientes, aunque disminuyó a largo plazo. Las especies restantes respondieron positivamente, NLD aparecía como variable significativamente predictora de su biomasa si se le aplicaba un retraso de 2 años, mientras que por sí sola mostró un poder explicativo bajo, excepto para la Focha común (Fulica atra). Al definir fases temporales homogéneas, los podicipédidos (Podiceps sp.) podrían ser identificados como especies de indicadoras de "alerta temprana" de eutrofización, y la Focha común como indicadora de etapas más avanzadas. El aumento de piscívoros lo largo de un período en el que las capturas de pescado disminuyeron podría reflejar un cambio en la composición o estructura de la comunidad de peces que favoreciera sus preferencias alimenticias. El papel interactivo de las medusas, amortiguando temporalmente la carga de nutrientes, también puede estar relacionado con estos cambios.

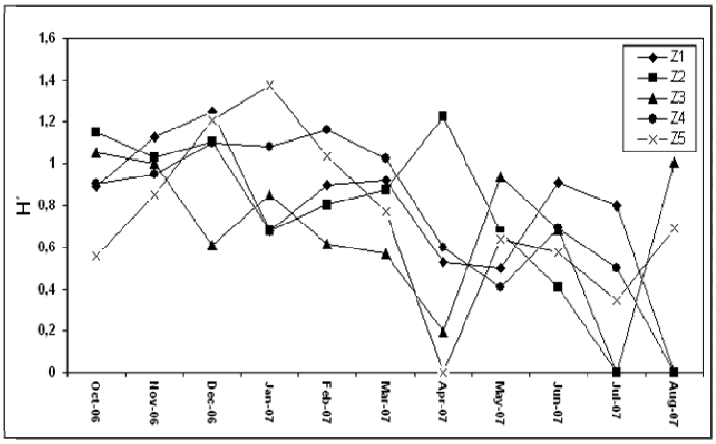
Referencia de publicación: Robledano, F., Esteve, M.A. Martínez-Fernández, J. and Farinós, P. 2011. Determinants of wintering waterbird changes in a Mediterranean coastal lagoon affected by eutrophication. Ecological Indicators, 11, 395-406.

Capítulo 2. Estructura y distribución de la comunidad de aves acuáticas en la alguna costera del Mar Menor (SE España) y su relación con gradientes ambientales

Resumen







De octubre de 2006 a marzo de 2008 se censó mensualmente la comunidad de aves acuáticas de la laguna costera del Mar Menor (SE España) con el fin de analizar la variación temporal y espacial de la abundancia y su relación con gradientes ambientales. En este muestreo se excluyó la fracción de la comunidad ligada al hábitat de ribera. El objetivo fue evaluar el papel de las aves acuáticas como indicadores

integrales, en relación a una zonificación representativa de la heterogeneidad ambiental de la laguna.

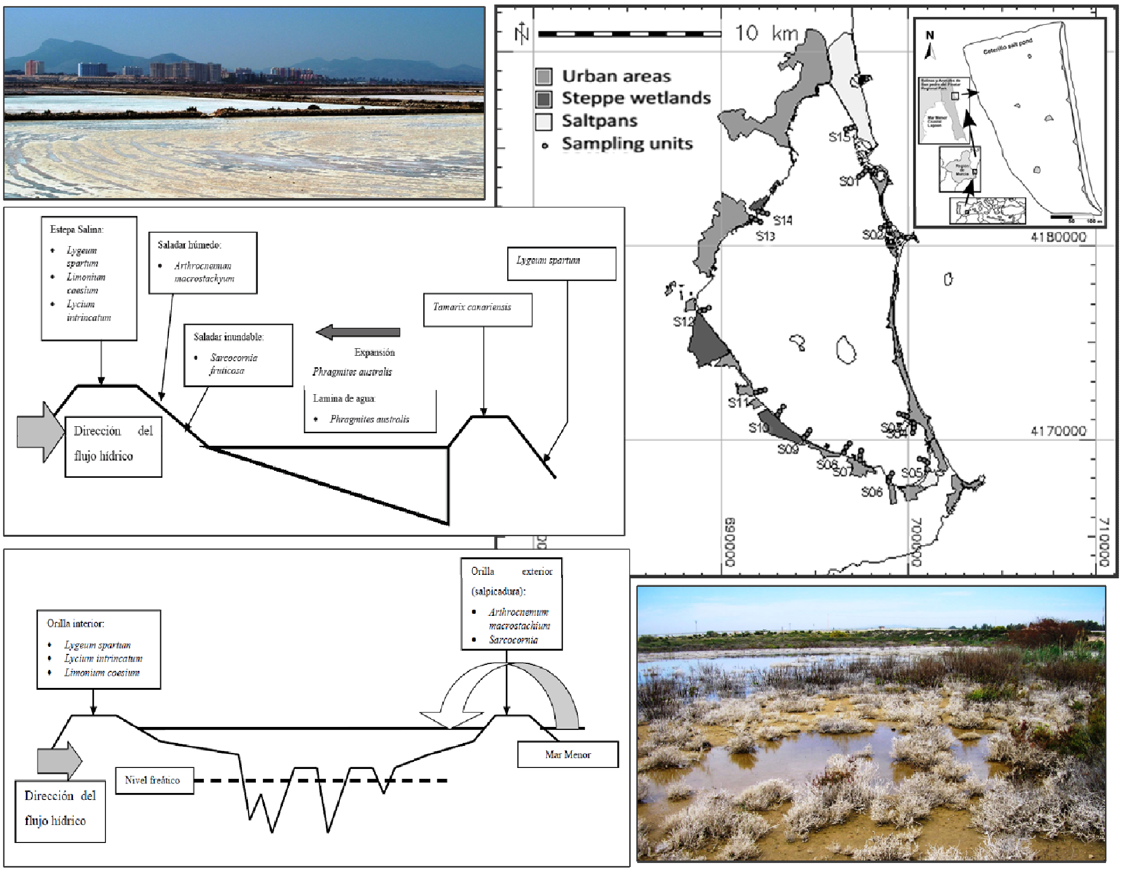
Las aves acuáticas se censaron en 20 estaciones de muestreo distribuidas de acuerdo a una zonificación previa y estandarizada, basada en el grado de confinamiento y de la productividad del sistema, y posteriormente modificada para ajustarse al uso que hacen las aves a escala de paisaje. Diferentes técnicas de clasificación y ordenación multivariante permitieron identificar especies indicadoras y mostrar su asociación con los vectores ambientales pertinentes. Gradientes opuestos de diversidad y abundancia se relacionan con un gradiente marino-continental, en el que la influencia mediterránea promueve la diversidad de aves y la influencia continental favorece la abundancia total. La abundancia también cambia en latitud, aumentando hacia el sur en invierno y viceversa en verano. Durante el invierno, la mitad sur de la laguna presenta la mayor abundancia y diversidad, con predominio de somormujos, gaviotas y cormoranes. Las zonas del norte muestran picos de abundancia y diversidad en verano, con predominio de gaviotas y charranes.

A nivel de especie, existe un uso diferente de la laguna en relación a la latitud, estando la mitad norte caracterizada por la presencia de Gaviota reidora (Larus ridibundus) y la mitad sur por el Somormujo lavanco (Podiceps cristatus) y el Cormorán grande (Phalacrocorax carbo). Aunque ciertas especies o gremios se asocian con zonas específicas (gaviotas y charranes con sectores del norte, piscívoros buceadores con sectores del sur, etc.), sus preferencias no pueden interpretarse únicamente en relación a características o procesos funcionales de la laguna, determinados elementos físicos y ciertos factores directos de influencia antrópica también deben ser tenidos en cuenta.

Referencia de publicación: Farinós, P. and Robledano, F. 2010. Structure and distribution of waterbird community in the Mar Menor coastal lagoon (SE, Spain) and relationships with environmental gradients. Waterbirds, 33(4), 479-493

Capítulo 3. Lagunas como hábitat de aves acuáticas: Respuestas de las comunidades al impacto humano y la gestión a lo largo de escalas espaciales y temporales

Resumen



Las lagunas costeras (CLS) son sistemas de transición que albergan notables valores y recursos biológicos al mismo tiempo que desempeñan papeles funcionales clave para los sistemas ecológicos y sociales. Las aves acuáticas son valiosos indicadores apicales de las condiciones del hábitat en las CLS. Los cambios en la intensidad de la influencia humana (en forma de impactos y/o de gestión) causan cambios dramáticos en la composición, la estructura y valor de conservación de sus comunidades de aves acuáticas, independientemente de las escalas temporales y espaciales en las que se examinen. Utilizamos los resultados de dos estudios llevados a cabo en un complejo lagunar mediterráneo del sureste de España (el Mar Menor), para

mostrar las diferentes escalas físicas y fases temporales sobre las que la comunidad de aves acuáticas cambia, utilizando diferentes parámetros e índices como señales de cambios ambientales con un influencia directa en la conservación de la biodiversidad.

Las zonas de estudio representan los extremos de un gradiente de tamaño: por un lado, una pequeña charca restaurada o sub-laguna (Charca del Coterillo) con unas pocas hectáreas de superficie, y por otro lado, la franja litoral de la laguna costera del Mar Menor (decenas de kilómetros cuadrados). Ambos sistemas comparten las características de ser aguas hipersalinas poco profundas con una fuerte dependencia de influencias externas, ya sean naturales (atmosféricas, hidrográficas) o artificiales (impactos hidrológicos, eutrofización, gestión). En particular, la falta de una gestión adecuada a escala de cuenca hidrográfica (en el caso de la laguna costera) o a escala local (en el caso de la pequeña charca) puede causar un deterioro del hábitat a largo plazo. Estos estudios complementan los realizados anteriormente en la parte más profunda de la cubeta principal de la laguna.

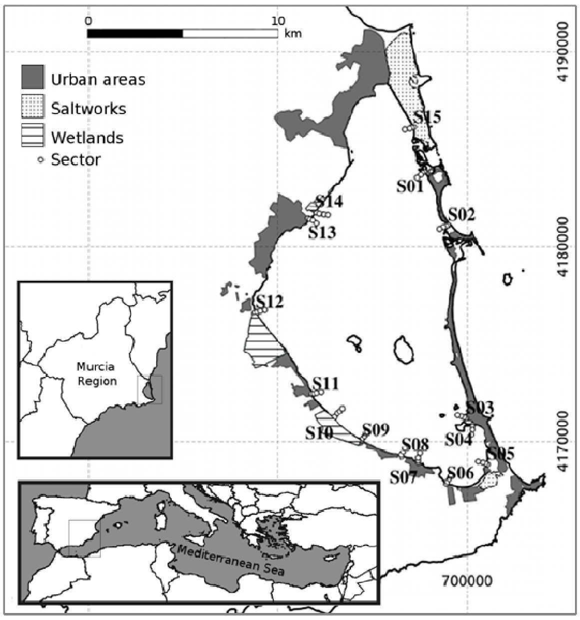
La comunidad de aves ha sido muestreada en ambos estudios a través de diferentes programas de censo. En el Coterillo, una charca salina anteriormente conectada con la industria salinera, se realizaron censos de la comunidad de aves acuáticas invernantes (mensuales) y nidificantes (anuales) entre 1999 y 2010. En el litoral de la laguna del Mar Menor, se censaron 15 unidades de muestreo entre 2006 y 2010, cubriendo dos periodos estivales (consideradas desde Abril a Septiembre) y tres periodos de invernada (Octubre-Marzo). La abundancia, riqueza, diversidad y determinados índices de conservación se utilizaron para describir las variaciones en la comunidad de aves acuáticas. Se realizaron análisis de similaridad (MDS) y de valor indicador (INDVAL) para identificar especies indicadoras de unidades espaciales y/o temporales (áreas o fases) con coherencia ecológica, y caracterizadas por condiciones derivadas de ciertos impactos o medidas de gestión.

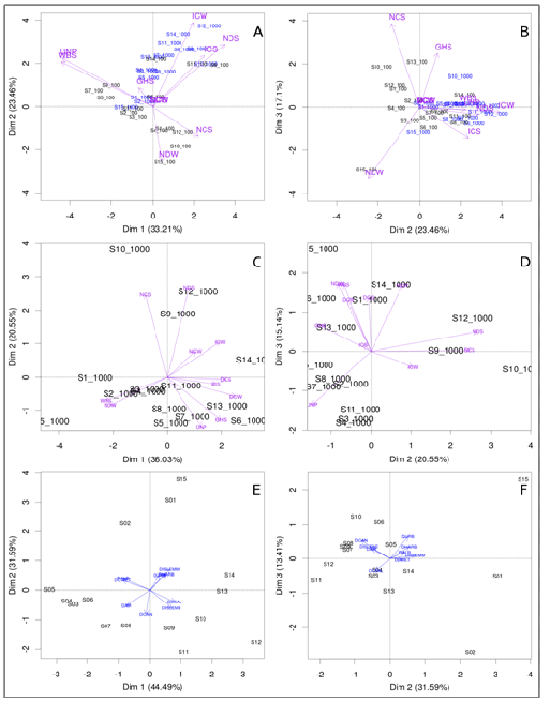
La persistencia de ciertos factores de amenaza de "fondo" (cambios en los usos del suelo o contaminación en la cuenca hidrográfica) y la falta de una monitorización continua, especialmente de las variables de impacto, hace difícil el aprender de los resultados de los estudios de aves acuáticas y aprovechar su valor indicador con fines de conservación. Nuestros resultados sugieren que, además de compensar la pérdida de hábitats naturales, la restauración de pequeñas lagunas o cubetas secundarias *constituye una potente herramienta para la gestión de los impactos en la cuenca vertiente. Además, las aves acuáticas representan un indicador clave del éxito de la gestión, tanto a escala local como escala de complejo de humedales y en relación tanto a la biodiversidad como al funcionamiento del ecosistema.*

Referencia de publicación: Farinós, P., Robledano, F., Perona, C. and Soto, A.J. 2013. Lagoons as a waterbird habitat: response of communities to human impact and management accross space and time scales. Mwinyihija, M. (Ed.). Lagoons: Habitats and Species, Human Impacts and Ecological Effects. Nova Science Publishers, Inc., NY. pp: 57-108.

Capítulo 4. Efectos antrópicos sobre los gradientes de paisaje y hábitat y su influencia sobre la comunidad de aves acuáticas en una laguna costera Mediterránea (Mar Menor, SE España)

Resumen





La evaluación de los efectos antropogénicos sobre el estado ecológico de las lagunas costeras del Mediterráneo, tal como exige la Directiva Marco del Agua (DMA), tiene que tomar en consideración a las aves acuáticas. Durante dos años completos (2006-2008) y dos temporadas de invernada adicionales (2009/10 y 10/11) la comunidad de aves acuáticas cercanas a la costa de la laguna costera del Mar Menor (SE, España) fue monitorizada mensualmente. Las comunidades de verano y de *invierno se estudiaron por separado. Los datos recogidos sobre aves acuáticas se analizaron en relación a gradientes ambientales que resumían la influencia estructural y funcional del paisaje circundante en diferentes escalas.*

En primer lugar, la variación espacial de los índices biológicos (uso total de las aves acuáticas, riqueza de especies, índice de diversidad de Shannon y especies indicadoras a nivel individual) mostró relación con los gradientes ambientales definidos previamente. La presencia de valores extremos de los índices en ciertas unidades espaciales de dichos gradientes, fue utilizada como criterio para la selección de variables respuesta (superficie ocupada por diferentes tipos de hábitats y usos del suelo, distancia a elementos atractivos y/o perturbadores en la orilla, etc.), que posteriormente fueron analizadas como variables predictoras de los índices de aves acuáticas.

La distancia a la costa tiene efecto sobre la estructura de la comunidad de aves acuáticas, destacando la importancia de la primera banda de 100 m (aguas someras), en el que muchas especies relevantes para la conservación convergen dada la existencia de recursos alimenticios accesibles, diversos y abundantes . El atractivo de los hábitats naturales (tramos de orilla bien conservados) también se refleja en los valores de riqueza y diversidad de la comunidad, así como en las preferencias de algunas especies (Garceta común Egretta garzetta, Gaviota picofina Larus genei y Charrancito Sterna albifrons). Esto apoya la necesidad de medidas de protección y restauración efectivas dirigidas a dichos hábitats, cuya conectividad con los paisajes interiores favorables resulta también esencial. También se ha detectado una influencia positiva de la conexión mediterránea (representado por el único canal funcional actualmente) sobre la diversidad de especies de aves acuáticas. Dicho efecto se relaciona con un gradiente de confinamiento-influencia marina (con un aumento de la diversidad de recursos alimenticios y nichos de alimentación cerca del mar abierto).

Dado que las respuestas específicas a procesos perturbadores locales (por ejemplo, la eutrofización, el estrés por actividades recreativas) aparecen ligadas a las preferencias de hábitat y paisaje, el valor de las aves acuáticas como señales ecológicas integrales en sistemas estuarinos, gana apoyo. Dentro de los hábitats y paisajes estructuralmente favorables, se espera que los índices basados en aves acuáticas respondan positivamente a las mejoras ambientales (por ejemplo, reducción *de la eutrofización, restauración de procesos funcionales de la laguna o regulación del aprovechamiento lúdico del agua), tanto en términos cualitativos como cuantitativos (riqueza de especies, diversidad, valor de conservación). Esto puede ayudar a armonizar los objetivos de la regulación de la calidad de las aguas (DMA) y la gestión y conservación de la biodiversidad (Directivas Aves y Hábitats de la UE).*

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Chapter 1. Determinants of wintering waterbird changes in a mediterranean coastal lagoon affected by eutrophication

Introduction

Wetlands and coastal waters experience strong pressures on their biodiversity and water quality (e.g. Maltby, 1986; Gibbs, 2000), due to the strong human demands on their surrounding land, shoreline and littoral waters (Gopal, 2003). Southern Mediterranean lagoons are among the most vulnerable systems to such pressures (De Stefano, 2004; Viaroli *et al*., 2005). They are influenced by intensively farmed and densely populated watersheds, particularly during summer when the Mediterranean becomes the major world's tourism hotspot (Vogiatzakis *et al*., 2006). Eutrophication is one of the main consequences (Caddy and Bakun, 1995), and despite a number of studies have focused on this process (e.g. Delbaere and Nieto-Serradilla, 2004; García Pintado *et al*., 2007), there is still little information about the response of biodiversity to the biotic changes that it brings. The lack of adequate datasets on nutrient inputs into the lagoons is an important obstacle for such research, although environmental modelling has partially offset this gap (Rodríguez *et al*., 2005). Unfortunately there are not comparable datasets on "response" biological variables for the same time span, with the exception of wintering waterbirds, regularly counted for decades in many Mediterranean wetlands (Hoffmann *et al*., 1996).

Waterbirds are considered useful indicators of changes in wetlands, responding in an integrative manner to stressors (Kushlan, 1993; Adamus, 1996; Amat and Green, 2009). Although the use of indicators based on bird numbers has been critiziced (Morrison, 1986; Landres *et al*., 1988), they often associate with changes in the structure and functioning of wetland ecosystems (Kerekes and Pollard, 1994; Noordhuis *et al*., 2002; Hebert *et al*., 2009). Besides, birds are well studied and monitored (many non-professional ornithologists are willing to help create data sets for large-scale studies). A growing number of papers are including waterbirds, coupled with other environmental data, in studies of water quality and ecological restoration of wetlands and coastal waters (Ysebaert *et al*., 2000; Stapanian *et al*., 2004; Holm and Clausen, 2006; Philippart *et al*., 2007).

In the framework of an EU-funded research project (DITTY), aimed at the development of decision-support tools for the management of Mediterranean lagoons and their watersheds, we started to consider waterbirds as indicators of environmental change. Wintering waterbird censuses in one of these lagoons showed a general positive response to simulations of nutrient input (Martínez *et al*., 2005; Amat and Green, 2009), a relationship already suggested by earlier studies warning about incipient eutrophication (Hernández and Robledano, 1997). Increases in turbidity, phytoplankton and filamentous algae, and lagoon-wide jellyfish blooms were symptoms of a further deterioration (Pérez-Ruzafa *et al*., 2002). The relationship of the main driver (nutrient load) with one of the most conspicuous biotic elements involved in this process (aquatic birds), is thus of special interest in the context of ecological quality management, and allows to relate biodiversity (bird species richness, abundance and community composition) and water quality indicators.

Here we analyse the relationship of waterbirds with locally measured or estimated environmental variables (nutrient load, fish production, jellyfish blooms), and discuss the value of birds, either as “early warners” of altered trophic stages, as monitors of changes during the eutrophication process, or as potential “performance indicators” of management. Specific objectives are: a) to assess the general response of the waterbird community to physical and biological variables describing local environmental changes as a result of agricultural and urban contamination; b) to assess waterbird response at the species or guild level, as a basis for the selection of bird metrics on which to monitor long-term changes in environmental conditions; and c) to discuss the bioindicator role of waterbirds, particularly as part of an early warning system, in the context of environmental and biotic interactions inside the lagoon.

Study area and methods

Study area

The Mar Menor Lagoon (135 km^2 surface area, 4 m mean depth) is a coastal lagoon of Southeastern Spain (Esteve and Martínez, 2003). It is separated from the Mediterranean Sea by a narrow sand strip almost completely reclaimed for tourism and surrounded by an irrigated agricultural plain of 480 km^2 inside a total watershed area of 1.275 km^2 (M.Francisca Carreño, personal communication) also with dense urban settlements (Figure 1). Ramsar Site since 1994, it is the largest lagoon of the western Mediterranean coast, and a remarkable biodiversity and scientific resource (EU Bird

Directive and Barcelona Convention's Specially Protected Area since 2001; see official information at http://liferayagri.carm.es/c/document_library/get_file?uuid=ef60e30c-2547-4e1d-a4ee-bf57af3ae65b&groupId=14).

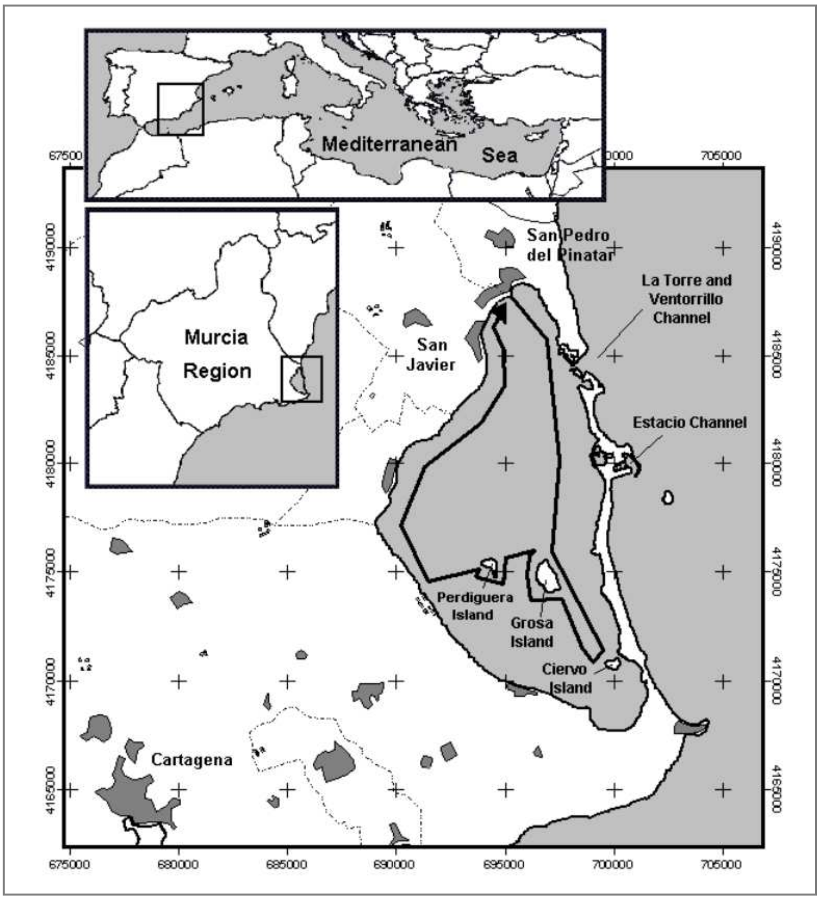

Figure 1. Map of the study area (Mar Menor Lagoon) with the boat itinerary used to survey waterbird populations inside the lagoon. Main municipalities surrounding the lagoon, islands within it and connections with the sea are shown.

It experienced major physical and hydrological changes in the decade of 1970's due to the dredging of one of its connections with the sea (Estacio channel). This increased the marine influence starting a process of "mediterraneization". Until mid 1980's the lagoon lacked any permanent incoming watercourse, but since then the channelized Albujón wadi started to discharge a permanent flow of agricultural drainage, refuse water from desalination and untreated urban sewage (García-Pintado *et al*., 2007). Total yearly flow has been estimated in 20 Hm^3 of water whose conductivity is usually between 8-11 mS/cm^2, a brackish input into the hypersaline (33.6-46.2 p.s.u) main water mass (Lloret *et al*., 2005). There are also low-salinity inputs through several minor channels, below-ground seepages and diffuse drainage.

Waterbird species and data

We used the January censuses made in the framework of the International Waterbird Census (IWC), compiled by Hernández and Robledano (1991), Martínez *et al*. (2005) and Hernández *et al*. (2006). 25 censuses were available between 1972 and 2005 (missing for 1974, 1976, 1980-82, 1998, 2000-01 and 2003). Waterbird numbers were recorded along a standardized boat route (Figure 1), and pooled by species for the whole lagoon. Five target species were selected, with abundance and feeding association to the lagoon as selection criteria: Great Crested Grebe *Podiceps cristatus*, Black-necked Grebe *Podiceps nigricollis,* Great Cormorant *Phalacrocorax carbo,* Red-breasted Merganser *Mergus serrator* and Common Coot *Fulica atra*; nomenclature according to Sibley and Monroe (1990). With the exception of Great Cormorant, which feeds in the lagoon but can forage in habitats outside it (irrigation ponds and the sea), most individuals of these species spend all their time budget inside the lagoon, feeding tipically in the water column and benthos. Wading birds, foraging in shoreline habitats of various depth, do not meet these criteria. Gulls and terns, numerically important in the bird assemblage, were not considered because of their looser trophic association with the lagoon, using external feeding habitats and anthropogenic sources of food unrelated to eutrophication.

The selected species, individually or at the family level, represent different foraging strategies (Table 1), even within the predominantly piscivorous guild. Yearly species abundances (total number in the January census) were converted into biomass multiplying by constant weight values (Cramp, 1980). The percent contribution of each species, family or guild to total biomass was used to illustrate changes in the trophic structure of the community.

Since local changes in waterbird numbers can obviously be linked to external influences, affecting their populations globally or on other geographical scales (Goss-Custard *et al*., 1995), there is a need to collect information on the status of wintering populations in higher-level biogeographical divisions (Lopes *et al*., 2005; Van Roomen *et al*., 2006). Regional level estimates were used as indicators of the general status of the populations in which the Mar Menor birds take part. Data were obtained from the project "Waterbird trends in Europe 1974-2002", as modelled total counts for the Western Mediterranean region (Wetlands International, 2008).

Familiy	**Species**	**Habitat specialization**	**Feeding specialization**	**Trophic guild**
Rallidae	Common Coot *Fulica atra*	Restricted (rarely outside lagoon)	Generalist	Herbivore
Phalacrocoracidae	Great Cormorant *Phalacrocorax carbo*	Oportunistic (moves around widely to terrestrial, aquatic and marine habitats)	Generalist	Piscivores
Podicipedidae	Great Crested Grebe *Podiceps cristatus*	Restricted (rarely outside lagoon)	Specialist	Piscivores
	Black-necked Grebe *Podiceps nigricollis*	Facultative (occasional movements to nearby wetlands)	Generalist	Piscivores
Anatidae	Red-breasted Merganser *Mergus serrater*	Restricted (rarely outside the lagoon)	Specialist	Piscivores

Table 1. Species studied classified "a priori" on the basis of their taxonomic, habitat and trophic characteristics. "Habitat specialization" refers to the degree of restriction of each species to the lagoon main water mass; "Feeding specialization" summarizes the variety of food types and foraging methods used by each species. The characterization is based on personal work in the area plus literature sources (Cramp, 1980). "Trophic guild" follows Ysebaert *et al.* (2000).

Nutrient load estimate

Total nitrogen load was estimated, on a yearly basis, from the watershed model developed for the lagoon, which integrates the key environmental and socio-economic factors driving the dynamics of nutrient inputs (Chapelle *et al.*, 2005; Martínez and Esteve, 2007; Martínez *et al.*, 2007). Due to the lack of data series on runoff for the entire watershed, a hydrological model has been used to estimate total runoff from rainfall and agricultural drainage (Chapelle *et al.*, 2005). The hydrological model is a physically-based, spatially distributed model which integrates an event-based approach (in case of rainfall episodes) within a continuous time approach. It generates long daily series of runoff, which constitute forcing inputs for the dynamic system model of the watershed, developed using the Vensim software (Ventana Systems, 2007). The estimated values for average annual total nitrogen loads in the Mar Menor watershed (1,045 ton/year in the period 2000-2005, with ample fluctuations ranging between 700-1,600 ton/year) fall within the ranges obtained in other intensive agricultural watersheds (Mattikalli and Richards, 1996; David *et al.*, 1997; Jordan *et al.*, 1997). They also agree with the scarce measured data on nitrogen concentration in the main flows of the Mar

Menor watershed (28-60 mg/l of DIN; Lloret *et al*., 2005; Velasco *et al*., 2006, Cabezas and Senent, 2009) and with the estimated average annual water discharges to the lagoon (37-45 Hm^3/year; Cabezas and Senent, 2009), which jointly provide an estimated annual nitrogen load into the lagoon between 1,000 and 1,300 ton/year. The volume of irrigation water has doubled between the last two decades, causing a consequent increase in the drainage flows into the lagoon and associated wetlands (Velasco *et al*., 2006; Carreño *et al*., 2008, Esteve *et al*., 2008, Martínez *et al*., 2009). Simulation results show an increasing trend in the input of nutrients from diffuse sources into the Mar Menor during these decades. This agrees with data on nitrogen concentration in the lagoon, which show a marked increase along the last two decades (Perez Ruzafa *et al*., 2002; Lloret *et al*., 2005; Velasco *et al*., 2006), reaching values ten times higher than those recorded at the end of the 1980s (Perez Ruzafa *et al*., 2002). Nutrient inflows show strong fluctuations due to the high variability in rainfall and the occurrence of flood events, as shown in other Mediterranean watersheds (David *et al*., 1997; Tournoud *et al*., 2003). This is also supported by the scarce empirical data on nitrogen concentration in the main ephemeral channels and in the lagoon during such events (Velasco *et al*., 2006, Cabezas and Senent, 2009).

Jellyfish data

In the late 1980s, the widening and deepening of the Estacio channel (Figure 1) moderated the lagoon’s temperature and salinity ranges, allowing two allochtonous species of *Scyphomedusae*, *Rhyzostoma pulmo* and *Cotylorhyza tuberculata,* to close their biological cycle inside the lagoon (Pérez Ruzafa and Aragón, 2003). In response to increased nutrient inputs generated by agricultural and tourism activities, summer proliferations began in the mid 1990s for both species, especially *C. tuberculata* (46 million individuals in summer 1997; Pérez-Ruzafa *et al*., 2002). Unpublished results of direct census, combined with data on annual catch and fishing effort (by fishermen committed to this task), provided by regional authorities and research institutions (Consejería de Agricultura y Agua, University of Murcia and Centro Oceanográfico del Mar Menor), allowed the estimation of the maximum summer population of jellyfish between 1988 (the starting date of the invasion) and 2004. Two variables were derived from this dataset: estimated total number of individuals reaching adult stage and a corrected figure after subtracting those estimated to have been removed.

Statistical analyses

Generalized Linear Models (GLMs) were used to examine relationships among variables, using waterbird biomass as dependent variable and the two environmental variables (nitrogen load and jellyfish number) as predictors (Table 2). We modelled the biomass of the five individual species (*Fulica atra* = BFUAT; *Podiceps cristatus* = BPOCR; *Podiceps nigricollis*= BPNIG; *Phalacrocorax carbo* = BPHCA; *Mergus serrator* = BMESE), and the aggregated biomass of non-monospecific families (*Podicipedidae* = BPODI) and feeding guilds (Piscivores = BPISC), as well as total bird biomass (BTWB). Predictor variables were the estimated total nitrogen load (NLD) during the preceding years (-1 yr or -2 yr), the estimated total number of jellyfish reaching adult stage during the corresponding growing seasons (JFS), and the corrected values for these after subtracting the estimated catch (JFSC). The same time lags (1-2 years) have been used in other dose-response studies of birds to nutrient changes in coastal waters (Philippart *et al*., 2007). We also used the annual catch of the two main lagoonal species exploited by the local fishing fleet (*Atherina boyeri* and *Engraulis encrasicholus*) as a surrogate index of fish productivity (FISH). These data were obtained from regional fisheries statistics (http://www.carm.es/econet/home.html). Although they refer to the whole year cycle, fishing activity for these species peaks in mid-winter when birds are counted (Alarcón, 2001). Bird biomass data were logarithmically transformed (log [x+1]). We used a Gaussian GLM model with identity link function, implemented through the freely-distributed software “R” (R Development Core Team, 2006). Prior to regression analysis, Pearson correlation coefficients among independent variables and between these and bird data were calculated as a previous step to facilitate the interpretation of the modelling output (Noordhuis *et al*., 2002).

Previous year(s) nutrient loads and the current fish catch were interpreted as surrogates of the productivity available for waterbirds staying in the lagoon in January, and jellyfish as the main top-down control on it (Pérez-Ruzafa *et al*., 2002; Pérez Ruzafa and Aragón, 2003) at an equal time scale. We tested models based in the expected influence of these variables on bird biomass through the following steps:

i) First, simple models tested the individual effect of nutrient input (NLD) and fish catch (FISH) on bird biomass variables.

ii) In a second step, multiple models for the combined effects of nutrient loads and fish catches with jellyfish on waterbird abundance were built adding new variables to the simple models of step 1. We searched for both additive and non-additive (interactive) effects. Significant additive effects of jellyfish variables (JFS or JFSC), nutrient effects remaining significant, would support some quantitative influence of gelatinous plankton upon the productivity exploitable by birds. Jellyfish can compete for nutrients with organisms that serve as food for waterbirds (see Vasas *et al*., 2007). On the other hand, significant interactions among variables could come from trophic effects different than direct competition for nutrients (e. g., shifts in prey composition). These changes would affect waterbird species or guilds in specific ways, since they display slightly different feeding preferences (feeding depths, prey size range…). Predictors were incorporated in the models following a stepwise procedure, according to the significance of coefficients and the change in deviance. A p-value based on the chi square test statistic was used to evaluate the significance of the reduction in deviance by each additional predictor incorporated to a model (McCullag and Nelder, 1989), and full models were compared on the basis of total explained deviance. Critical P-level was set at 0.2 in order to retain the influence of relevant environmental predictors on some response variables (Noordhuis *et al*., 2002).

To deal with the potential effect of external factors, prior to testing environmental variables, we regressed the biomass of each species against its Western Mediterranean population index (see *Waterbird species and data* section), considering this as a first determinant of local wintering populations when significative relationships emerged. These data were available for the period 1979-2002 (Red-breasted Merganser), 1989-2002 (Great Cormorant and Great Crested Grebe) and 1993-2002 (Common Coot). For this reason, correlation and regression analyses were performed on two periods (reduced dataset corresponding to the years with population index for each species). As in the previous procedure, we first tested for the influence of the index alone and added environmental variables for these same time series if significant relationships appeared. No data were available for Black-necked Grebe. We also gathered bibliographic information on winter census or trends of the five species studied at other geographical/administrative scales: whole Spain (Martí and Del Moral, 2003) and Valencian Community, the nearest autonomous region (Gómez *et al*., 2006), using it to compare trends and discuss the potential effects of local versus external factors on Mar Menor waterbird numbers.

Results

Nutrients, jellyfish and fish catches

Nitrogen load showed strong interannual fluctuations, but two main phases of change were apparent. After a slow increase from 1973 to 1994, a much steeper but fluctuant rise occurred during 1995-2003. Total jellyfish exhibited a typical bloom response. They showed a steep increase from 1992, peaking in 2001 and dropping rapidly to values near those of 1999 by 2003 (Figure 2, consider the 2-year lag). Catch reduced total jellyfish number by up to 45.4% (2003 campaign) and on average 12.5%. Fish catches experienced an overall decline, although some recovery and a latter stabilization occurred after 1999 (Figure 2).

Waterbird numbers and community composition

Of the five species studied, all but Red-breasted Merganser have increased in numbers along most of the period of study. Among the trajectories of the three piscivores that increase, the greatest fluctuations (and the highest counts) correspond to Black-necked Grebe. The trend of the Great Crested Grebe is parallel to that of its congeneric species, but less fluctuant. Great Cormorant numbers seem to have grown more steadily. Coot display also a marked increase, but starting later than the three increasing piscivores that show this same trend (Figure 3).

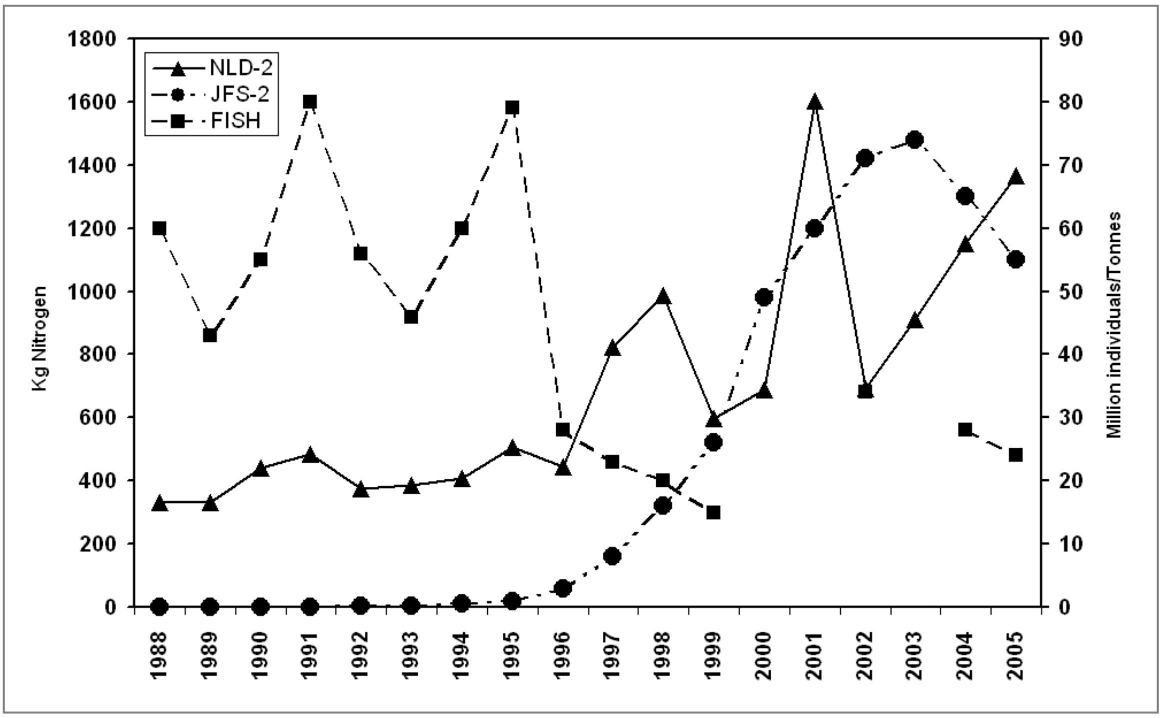

Figure 2. Estimates of total nitrogen load (NLD-2) and total jellyfish number (JFS-2) in the Mar Menor Lagoon (1989-2005). both represented with a 2-year lag with respect to the January waterbird census (dependent variable in regression analysis). Annual fish catch from the lagoon (not lagged) is also shown. Nitrogen (in Tonnes) is represented in the primary y axis; fish catch (in Tonnes) and jellyfish number (million individuals) in the secondary one

Figure 4 summarizes the general progression of waterbird biomass and the relative contribution of each species. Although there are gaps at the start and the end of the study period, there are recognizable periods characterised by the dominance of particular species. Red-breasted Merganser is the dominant piscivore during most of the decade of 1970. Great Cormorant dominates the community most years since then, representing ca. 50% of the biomass except between 1988 and 1996 when other piscivores, including the two grebe species, dominate. Herbivores (coot) join the community in 1992 and their contribution rises to more than 30% in 2005. The biomass of Red-breasted does not change markedly in the long term, although its relative contribution decreases gradually (from more than 90% to less than 15%).

In terms of feeding guild abundance, the composition of the lagoon’s waterbird community has changed from a 100% of piscivores until 1991, to nearly 50% of herbivores in 2005. When expressed as biomass, the contribution of herbivores is slightly lower (under 40%). In this last year the ratio herbivores/piscivores was 0.83 (in numbers) and 0.58 (in biomass).

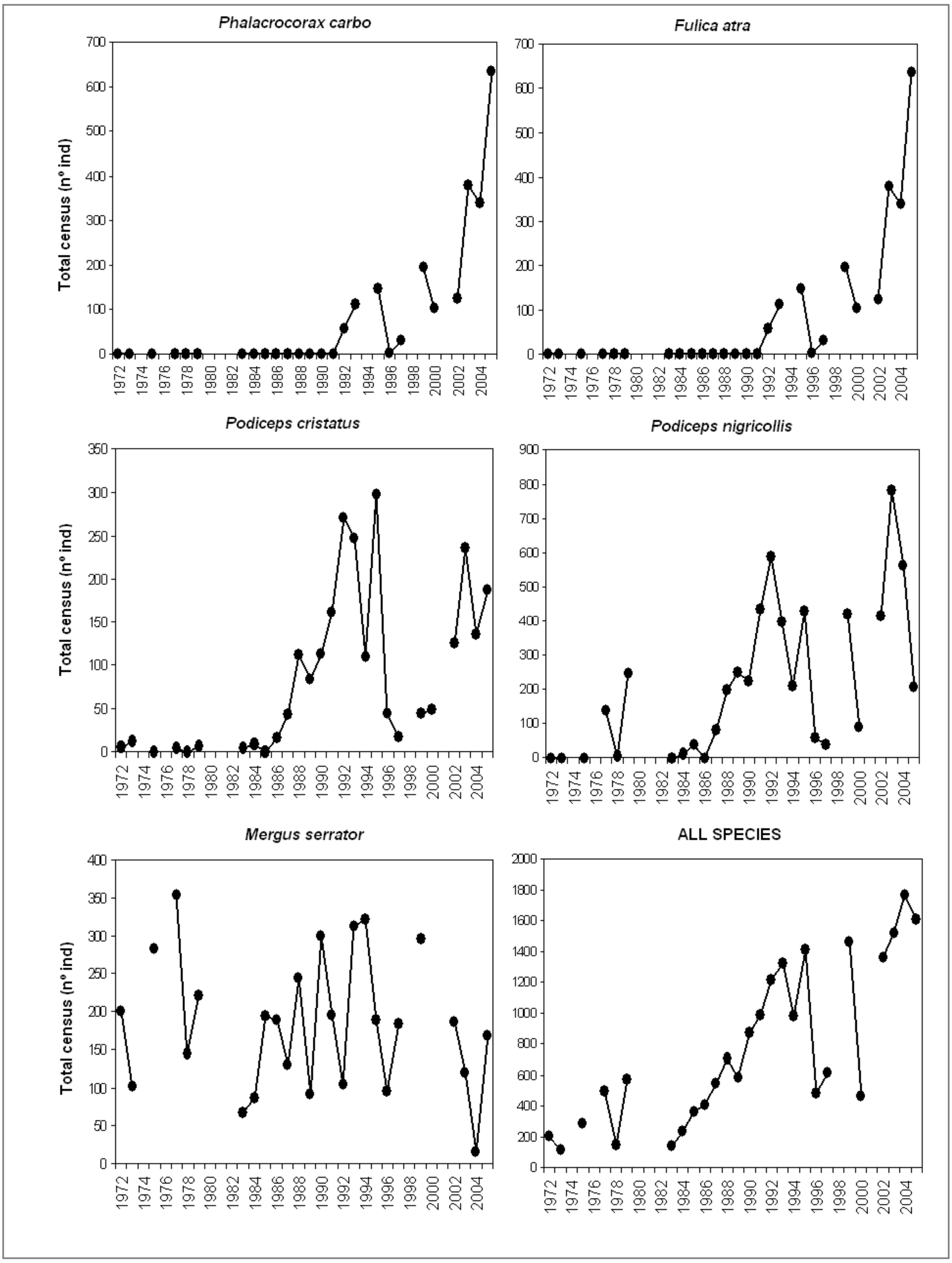

Figure 3. Results of the January waterbird census of the five species studied in the Mar Menor Lagoon (total number of individuals counted in the boat survey)

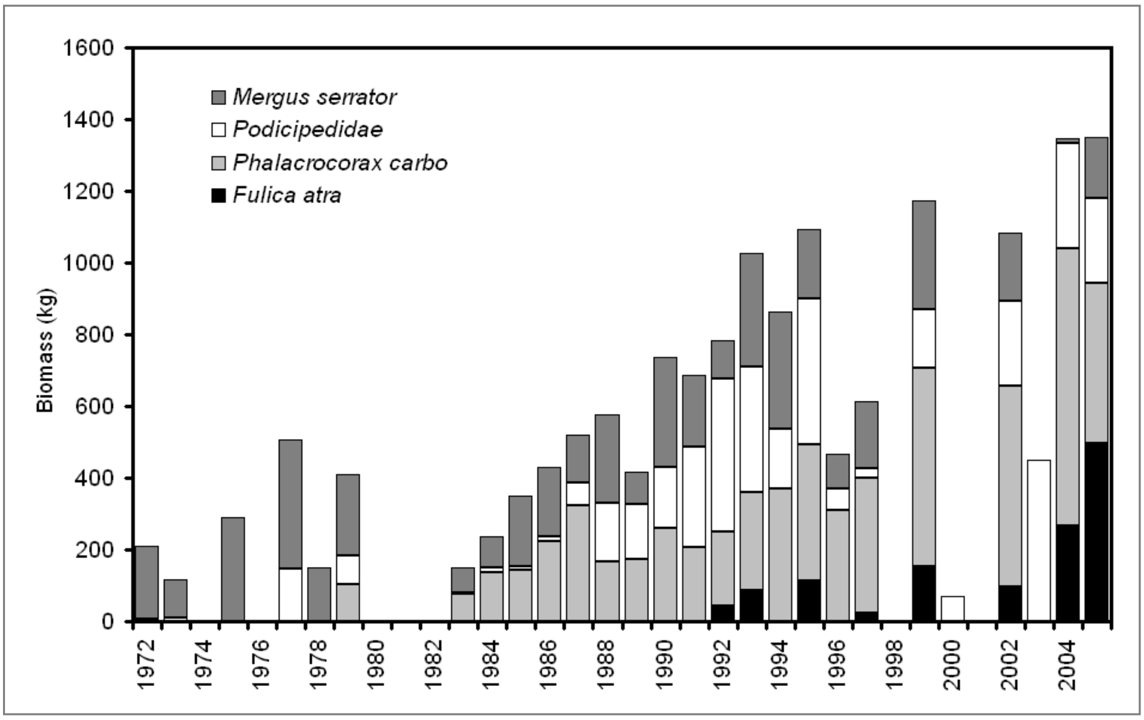

Figure 4. Contribution of the five species studied to the total biomass (kg) of the waterbird community of the Mar Menor Lagoon (only years with data for all the species altogether are shown).

Waterbird response models

Pearson correlation coefficients showed significant relationships among Nitrogen load and jellyfish variables in most cases. Between 1989 and 2002 significant correlations where found between nitrogen load and uncorrected jellyfish number when one year lag was considered, and with both corrected and uncorrected figures when two. Significant negative correlations were also found between FISH and all jellyfish variables, with coefficients ranging from 0.74 and 0.78. The biomass of *Podiceps cristatus* was negatively correlated with some jellyfish variables and positively with FISH. On the opposite, *Phalacrocorax carbo* was positively associated with jellyfish (and nutrient) variables (Table 2).

The full dataset produced strong and significative correlations among nutrient and jellyfish variables, all exceeding the 0.7 threshold. FISH correlated negatively and significantly with all the environmental variables but coefficients were always below 0.6. The biomass of Common Coot and Great Cormorant were positively correlated with nutrient variables, particularly when a 2-year lag was considered. Great Cormorant biomass was negatively correlated with FISH (r = -0.579) (Table 2)

1989-2002 (1993-2002 for *Fulica atra*; 1979-2002 for *Mergus serrator*)														
	NLD-1	(p)	NLD-2	(p)	JFS-1	(p)	JFSC-1	(p)	JFS-2	(p)	JFSC-2	(p)	FISH	(p)
NLD-1	-		-											
NLD-2	-		-											
JFS-1	0.651	*												
JFSC-1	0.521	ns												
JFS-2	-		0.680	**	-									
JFSC-2	-		0.597	*	-									
FISH	-0.495	ns	-0.348	ns	-0.784	***	-0.752	**	-0.740	***	-0.776	**		
BFUAT	0.112	ns	0.429	ns	0.291	ns	0.006	ns	0.498	ns	0.312	ns	-0.040	ns
BPOCR	-0.462	ns	0.033	ns	-0.591	*	-0.565	ns	-0.566	ns	-0.595	*	0.723	**
BPHCA	0.599	*	0.713	**	0.660	*	0.650	*	0.630	*	0.591	*	-0.419	ns
BMESE	0.284	ns	0.183	ns	0.165	ns	0.157	ns	0.192	ns	0.207	ns	-0.018	ns
Full dataset														
	NLD-1	(p)	NLD-2	(p)	JFS-1	(p)	JFSC-1	(p)	JFS-2	(p)	JFSC-2	(p)	FISH	(p)
NLD-1														
NLD-2														
JFS-1	0.812	***												
JFSC-1	0.744	***												
JFS-2			0.803	***										
JFSC-2			0.755	***										
FISH	-0.595	***	-0.526	**	-0.580	**	-0.570	**	-0.535	**	-0.531	**		
BFUAT	0.704	***	0.837	***	0.731	***	0.600	***	0.785	***	0.670	***	-0.482	***
BPOCR	0.244	ns	0.386	ns	0.237	ns	0.144	ns	0.292	ns	0.244	ns	-0.119	ns
BPNIG	0.270	ns	0.390	*	0.435	*	0.339	ns	0.470	ns	0.444	*	-0.247	ns
BPHCA	0.650	***	0.749	***	0.715	***	0.648	***	0.715	***	0.654	***	-0.579	**
BMESE	-0.300	ns	-0.226	ns	-0.174	ns	-0.086	ns	-0.279	ns	-0.205	ns	0.155	ns
BPODI	0.273	ns	0.417	*	0.346	ns	0.244	ns	0.396	ns	0.355	ns	-0.187	ns
BPISC	0.352	ns	0.532	**	0.428	*	0.394	*	0.400	*	0.355	ns	-0.370	ns
BTWB	0.509	**	0.717	***	0.574	**	0.497	**	0.576	**	0.499	*	-0.452	*

Table 2. Correlation coefficients among environmental variables and between these and the biomass of waterbird species and guilds. JFS-1 = Total Jellyfish (previous year); JFSC-1 = id., corrected for captures; JFS-2 = Total Jellyfish (2 yr lag); JFSC-2 = id., corrected for captures; FISH = Fish catch; BFUAT = Biomass of *Fulica atra*; BPOCR = Biomass of *Podiceps cristatus*; BPNIG = Biomass of *Podiceps nigricollis*; BPHCA = Biomass of *Phalacrocorax carbo*; BMESE = Biomass of *Mergus serrator*; BPODI = Biomass of *Podicipedidae*; BPISC = Biomass of piscivores; BTWB = Total waterbird biomass. Significance codes: (***) <0.001, (**) <0.01 (*) < 0.05.

In the first step of GLMs, the Western Mediterranean Index of wintering population (WMPI) was a significant predictor of local abundance only for Great Cormorant (Table 3), explaining almost 50% of the variation in local biomass. Considering the same period (1989-2002), Nitrogen load (NLD), whichever lag considered, individually explained similar proportions of deviance (40-50%). The model with WMPI and NLD (2-yr lag) as predictors was the only significant one that included more than a single variable, explaining 58.21% of deviance. When the full dataset (1972-2005) was taken into account, nutrient loads individually predicted between 15-20% of the variation in waterbird biomass, but the best predictor was FISH with a negative coefficient (Table 3).

Dependent variable: Biomass of *Phalacrocorax carbo* (BPHCA)

Predictor variables:	**Period**	**Coefficient (β)**	**SE**	**P**	**Deviance explained (%)**
WMPI	1989-2002	4.07x10^{-6}	1.32x10^{-6}	<0.05*	48.54
+ NLD (-2 yr)		5.5x10-4	3.81x10-4	0.18	58.21
NLD (-1 yr)	1989-2002	0.0006	0.0002	<0.05*	39.75
NLD (-2 yr)	1989-2002	0.0008	0.0002	<0.01**	50.79
NLD (-1 yr)	1972-2005	0.0021	0.0010	0.054	15.09
NLD (-2 yr)	1972-2005	0.0031	0.0013	<0.05*	20.18
FISH	1972-2005	-0.0450	0.0163	<0.05*	28.04

Dependent variable: Biomass of *Fulica atra* (BFUAT)

Predictor variables:	**Period**	**Coefficient (β)**	**SE**	**P**	**Dev expl (%)**
NLD (-1 yr)	1972-2005	0.0047	0.0009	<0.001***	53.69
+ NLD (-1 yr)*JFSC (-1 yr)		-8.7x10-5	5.36x10-5	0.11	67.10
NLD (-2 yr)	1972-2005	0.0054	0.0009	<0.001***	59.59
+ NLD (-2 yr)*JFS (-2 yr)		-1.11 x10-4	5.22x10-5	<0.05*	71.87
+ JFSC (-2 yr)		0.0469	0.0219	<0.05*	65.53
+ NLD (-2 yr)*JFSC (-2 yr)		-1.24 x10-4	5.43x10-5	<0.05*	72.19
FISH	1972-2005	-0.0529	0.0161	<0.01**	31.81

Dependent variable: Biomass of Podiceps cristatus (BPOCR)

Predictor variables:	**Period**	**Coefficient (β)**	**SE**	**P**	**Dev expl (%)**
NLD (-1 yr)	1972-2005	0.0018	0.0008	<0.05*	16.38
NLD (-2 yr)	1972-2005	0.0029	0.0011	<0.05*	21.90

Dependent variable: Biomass of *Podiceps nigricollis* (BPNIG)

Predictor variables:	**Period**	**Coefficient (β)**	**SE**	**P**	**Dev expl (%)**
NLD (-1 yr)	1972-2005	0.0015	0.0009	0.09	10.49
NLD (-2 yr)	1972-2005	0.0026	0.0012	<0.05*	16.58
FISH	1972-2005	-0.0306	0.0130	<0.05*	19.97

Dependent variable: Biomass of *Mergus serrator* (BMESE)					
Predictor variables:	**Period**	**Coefficient (β)**	**SE**	**P**	**Dev expl (%)**
NLD (-1 yr)	1972-2005	-0.0007	0.0003	<0.05*	15.95
NLD (-2 yr)	1972-2005	-0.0007	0.0004	0.11	10.24

Table 3. Waterbird individual response models. Only predictors with significative coefficients at p<0.2 were included according to percent deviance explained, and full models compared on the basis of total deviance explained. Predictor abbreviations like in Table 2, plus West Mediterranean Population Index (WMPI). Models including interactions imply that the variable also entered the additive model with lower explanatory power. Significance codes like in Table 2.

The models for Common Coot included only nitrogen load (1-yr lag) and all the jellyfish variables as single predictors. Nutrient load alone explained nearly 60% of waterbird variation. Jellyfish did not affect negatively the biomass of waterbirds. However, the model including the interaction between NLD and corrected jellyfish number (2-yr lag) with a negative coefficient, explained up to a 72.19% of deviance (Table 3).

NLD (1- or 2-yr lag) were the only significant predictors of *Podiceps cristatus* biomass, explaining up to 21.9% of deviance. Black-necked Grebe individual models, in turn, included NLD (2-yr lag) and FISH as significant predictors, with opposite signs. NLD explained 16.58% of waterbird variation, and the negative relationship with FISH explained 19.97% of it (Table 3).

NLD (1-yr lag) was also the only significant predictor of Red-breasted Merganser biomass, explaining 15.95% of deviance, but this variable had a negative effect on the dependent variable (Table 3).

When the total biomass of *Podicipedidae* was used as dependent variable, NLD (2-yr lag) was the only significant predictor, explaining 18.83 of deviance. FISH had a negative effect, and the interaction among these two variables was a positive significant predictor of grebe biomass, explaining 32.2 % of its variation (Table 4).

Dependent variable: Biomass of *Podicipedidae*

Predictor variables:	**Period**	**Coefficient (β)**	**SE**	**P**	**Dev expl (%)**
NLD (-1 yr)	1972-2005	0.0015	0.0008	0.06	12.65
NLD (-2 yr)	1972-2005	0.0026	0.0011	<0.05*	18.83
FISH	1972-2005	-0.0185	0.0126	0.15	8.95
+ FISH*NLD (-2 yr)		2.18x10-4	9.28x10-5	<0.05*	33.20

Dependent variable: Biomass of all piscivores

Predictor variables:	**Period**	**Coefficient (β)**	**SE**	**P**	**Dev expl (%)**
NLD (-1 yr)	1972-2005	0.0005	0.0003	0.09	10.78
+NLD(-1 yr)*JFSC (-1 yr)		-4.51x10-5	2.13x10-5	<0.05*	26.62
NLD (-2 yr)	1972-2005	0.0011	0.0004	<0.05*	24.34
+NLD(-2 yr)*JFSC (-2 yr)		-5.55x10-5	3.75x10-5	0.15	30.10
FISH	1972-2005	-0.0094	0.0043	<0.05*	17.91
+FISH*NLD (-2 yr)		8.91x10-4	2.99x10-5	<0.01**	46.86

Dependent variable: Total waterbird biomass

Predictor variables:	**Period**	**Coefficient (β)**	**SE**	**P**	**Dev expl (%)**
NLD (-1 yr)	1972-2005	0.0007	0.0003	<0.05*	19.84
+NLD (-1 yr)*JFSC (-1 yr)		-4.65x10-5	2.21x10-5	<0.05*	33.71
NLD (-2 yr)	1972-2005	0.0014	0.0003	<0.01**	37.70
FISH	1972-2005	-0.0119	0.0045	<0.05*	23.73

Table 4. Guild or total waterbird community response models. Only predictors with significative coefficients at p<0.2 were included according to percent deviance explained and full models compared on the basis of total deviance explained. Abbreviations like in Table 3, and significance codes like in Table 2.

A similar result was found with total piscivore biomass as dependent variable, although in this case NLD-2yr (positive) and FISH (negative) were both significant single predictor. The interactions NLD-2yr*JFSC-2yr (with a negative coefficient) and FISH*NLD-2yr (with a positive one) were also significant, with up to 46.86% of deviance explained by the model including the latter (Table 4). When the biomass of all piscivores minus Great Cormorant was used as response variable, no significant effect was found, nor of NLD (whatever the lag considered) and FISH (p>0.2).

Total community biomass was significantly and positively related with all the nutrient variables, and negatively with FISH. The model with NLD-2yr as single predictor explained the highest proportion of deviance (37.7%). The interaction NLD-1yr*JFSC-1yr was also a significant predictor and reversed the sign of the effect of these variables when considered individually (Table 4).

Discussion

Intensive agriculture and residential tourism have become the main pressures driving environmental change within the Mar Menor Lagoon during the last three decades. Total waterbird biomass increased more than 4-fold during this period. The increase in carrying capacity is a plausible explanation for this response (Ysebaert, 2000; Van Eerden *et al.*, 2005; Van Roomen *et al.*, 2006), supported by the lack of a general relationship with biogeographical population trends. Only the biomass of Great Cormorant, the species less tightly tied to the lagoon and less specialized in terms of foraging strategy, is predicted significantly by its Western Mediterranean population index. Cormorants respond positively to eutrophication, but they seem to do so at any scale considered. Local changes in most species also differ markedly from other nearby sites or regions. As shown in Martínez *et al*. (2005), even when trends showed the same direction than the Spanish populations (Martí and Del Moral, 2003), the shape or slope of the curves differed greatly. At a regional level, in some species we found opposite -rather than divergent- trends. This is the case of Common Coot, which in the Valencian Community (Gómez *et al*., 2006) declined sharply from 1991 onwards (from more than 14,000 wintering birds to less than 2,000), while undergoing a 10-fold increase in the Mar Menor between 1992 and 2005. In the same period, Great Crested Grebe showed a stable or slightly increasing trend in the Valencian Community, with on average 113 wintering birds, while in the Mar Menor it averaged 111, meaning that this hypersaline lagoon alone sustained a population equal to all the Valencian census. The carrying capacity of the lagoon for this phytophagous species seemed to increase during the second half of that period, opposite to what happened in a number of apparently more suitable Valencian wetlands. Black-necked Grebe was the only species with similar trends in the Valencian community and the Mar Menor, although peak counts did not coincide exactly.

Thus, nutrient load may be invoked as the first determinant of waterbird biomass in three out of five species, with the Great Cormorant population primarily governed by external factors and the Red-breasted Merganser apparently indifferent (tolerant) to nutrient loading during most of the study period, but negatively affected in the long term (Table 3; see also Martínez *et al.*, 2005). These two species should therefore be considered as poorer signals of lagoon deterioration. Although the models included significant effects of nutrient load for most species, their predictive value was limited (higher usually when a 2-year lag was allowed). Over a shorter time span nutrient load apparently explained a greater proportion of the variation in Great Cormorant biomass (up to 50.7%), but when considering the full dataset this was lowered to 15-20%. Along the shorter period, most of the variation was already explained by the Western Mediterranean population index alone. The deviance explained by nitrogen load for the full dataset would represent the true effect of lagoon eutrophication on this species. When the model was built using the biomass of all piscivores excluding Great Cormorant, no significant effect was found, which supports the strong bias exerted by this species on the global response. Along the extended period, individual models for all species had a similar explanatory power (10.24-21.90% deviance) except for Common Coot (up to 59.5%) (Table 3). But when pooled together, the global picture seems to be blurred by the differential response of each species or family to nutrient enrichment (Noordhuis *et al.*, 2002; Rönkä *et al.*, 2005).

The low predictive capacity of nitrogen load is not unexpected. Other variables could indicate best the trophic resources exploitable for waterbirds than the gross descriptors we have used as predictors. Apart from agricultural nitrogen, changes in waterbird abundance can be explained by: phosphorus loads from urban sewage (García-Pintado *et al.*, 2007); climatic factors (Gili and Pagès, 2005; Philippart *et al.*, 2007); salinity changes (Rosique, 2000; Hernández and Robledano, 1997); weather and disturbance (Davidson and Rothwell, 1993; Rönkä *et al.*, 2005; O'Connell *et al.*, 2007). Also, variations in the winter pattern of occurrence of the species, related to some of these factors or stochastic in nature (Lopes *et al.*, 2005), could explain apparent interannual changes (Van Roomen *et al.*, 2006). Long term changes in several geomorphological and hydrographic parameters, through its effect on fish community composition and abundance (Pérez Ruzafa *et al.*, 2007), can also influence waterbird numbers, but the lack of comparable abiotic and biological data series hamper their incorporation into the models (Gili and Pagès, 2005).

The temporal sequence of years can be split into phases of relative stability and quick transitions in the lagoon's ecological status. A similar -and somewhat parallel approach- has been applied to terrestrial birds in steppic wetlands surrounding the lagoon (Robledano *et al*., 2010). Within the general waterbird biomass increase that starts by mid 1980's, we can differentiate four main phases plus a short period of rapid change (Figure 5 and Table 5): Phase 1 and 2 correspond to a period of more or less homogeneous nutrient loading, the first part of which (1972-79) still shows high fish catches, but decreasing dramatically at its end (probably due to overfishing); the second phase (1980-87) shows a first positive response of generalist piscivores, mainly Great Cormorant, which dominates the community with Red-breasted Merganser. It follows a period of gradually increasing nutrient loads extending approximately from 1988 to 1995 (Phase 3), during which the relative biomass of *Podicipedidae* (vs that of Cormorant+Merganser) increases markedly; this is followed by a short phase (1996-98) with low fish catches, higher nutrient loads and an incipient jellyfish development (Phase 4), coincident with a fall in *Podicipedidae* biomass; and from 1999 onwards, there is some recovery of *Podicipedidae* but their relative contribution to total biomass is lowered due to the large increase of Great Cormorant and the also increasing share of herbivores (Common Coot). This last period (Phase 5) is characterized by further increases in nutrient loadings, but also by a greater abundance of jellyfish that could exert some control on eutrophication. In fact there is an apparent recovery of fish catches that can also be related to such control, which in turn could be responsible for the recovery of piscivores after 1998.

This sequence provides a good framework to discuss the bioindicator role of waterbirds, particularly as part of an alert system. Grebes can be identified as early warners of eutrophication, and Common Coot as late-stage indicators. It has been assumed that eutrophication benefits these two species (see Fjeldså, 2004; Rönkä *et al*., 2005, and references therein), but the response of waterbirds changes over time and among taxa depending on the stage or phase within this process (Van Impe, 1985; Hoyer and Canfield, 1994; Raffaelli, 1999; Van Eerden *et al*., 2005; Rönkä *et al*., 2005). Therefore, it is important to take advantage of the early warning function of grebes over a period of relatively unnoticed change in trophic status. This period is also the phase of higher community diversity (expressed here by the share of total biomass), a characteristic feature of waterbird assemblages of moderately eutrophic waters (Fjeldså, 2004; Rönkä *et al.*, 2005; MacDonald, 2006). This apparently beneficial effect could delay the adoption of measures to remedy the inflow of nutrients.

	Phase 1	**Phase 2**	**Phase 3**	**Phase 4**	**Phase 5**
	1972-79	**1980-87**	**1988-1995**	**1996-(97)-?**	**?-(1999)-2005**
NLD	Start and stabilization of nutrient inputs	Regular. homogeneous nutrient loading	Moderate increase in nutrient input	Sharp increase in nutrient input	Higher nutrient loading (fluctuant)
FISH	High fishing yield but declining at the end of period (overfishing?)	Very few data on fishing yields (probably fluctuant at intermediate levels)	Fishing yield fluctuant at intermediate levels	Sharp decline in fishing yields	Fishing yields stable or fluctuant at low levels
JFS	Absent	Absent	Absent	Incipient populations	Dramatic increase and peak numbers
Water birds	Dominance of Mergus serrator Low piscivore diversity	Dominance of *Phalacrocorax carbo* + *Mergus serrator*	Increase of *Podicipedidae* (maximum relative contribution) Maximum piscivore diversity	Decrease of *Podicipedidae* Dominance of *Phalacrocorax*	Increase of herbivores (*Fulica*) Partial recovery of *Podicipedidae*

Table 5. Characteristics of the main phases identified by the response of waterbirds to the environmental variables represented in Figure 5.

Towards the end of the study period, the continued increase in nutrient loadings and the decline of jellyfish numbers could illustrate a new shift, in this case towards conditions more favourable for herbivores. Eutrophication causes a marked deterioration of seagrass and macroalgal communities, apparently not beneficial for herbivores (Noordhuis *et al.*, 2002), but it also can favour opportunistic macroalgae (Krause-Jensen *et al.*, 2008), a source of food for generalist herbivores like Common Coot (Perrow *et al.*, 1997; Yallop *et al.*, 2004). The proliferation of such algae is already observed in some stretches of the lagoon’s shoreline. At this final stage, the lagoon still seems to hold a relatively diverse waterbird community, but there has been a major change in composition and structure with respect to the original community of hypersaline and oligotrophic waters. Conditions for piscivores, the earlier indicators of trophic change, could also deteriorate in this case. The feeding activity of Great Crested Grebe in eutrophic waters has been shown to be limited by factors like underwater visibility, exploitable fish biomass, fish size, and vertical distribution of the prey (Gwiazda, 1993; Van Eerden *et al*., 2003). The Great Cormorant, also, has been proposed as an indicator of waters of intermediate turbidity, above (or below) which it feeds less efficiently (Van Rijn and Van Eerden, 2003). Moreover, the favourable conditions for herbivores could become worse if the system shifts to a phytoplankton-dominated state (Noordhuis *et al*., 2002; Garcia-Pintado *et al*., 2007). If internal gradients in these conditions occur inside the lagoon, attention should also be paid to the numerical and distributional changes of piscivores and herbivores to anticipate the expansion of eutrophication symptoms within it (Robledano *et al*., 2008).

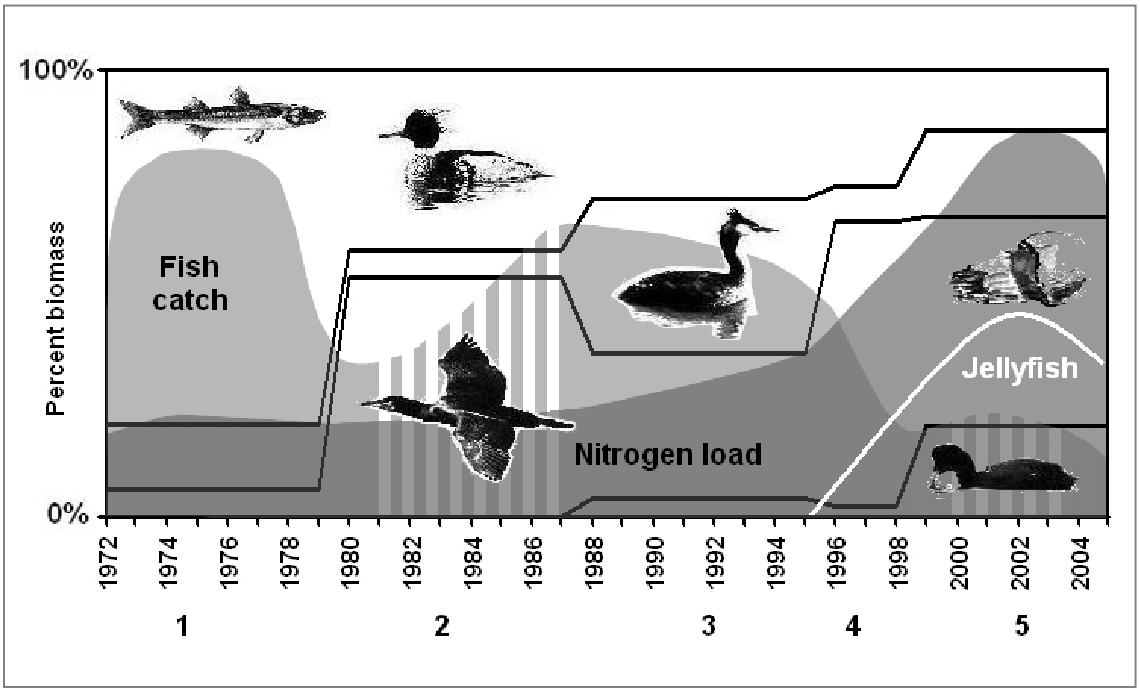

Figure 5. Graphic representation of the main phases identified by the response of waterbirds to environmental variables. Waterbirds are represented by areas proportional to their average contribution to total community biomass (from top to bottom: *Mergus serrator*, *Podicipedidae*, *Phalacrocorax carbo*, *Fulica atra*) and environmental variables by curves or areas (simplified from figure 2; light grey = fish catch; dark grey = Nitrogen load; white line = jellyfish numbers). Hatched areas are periods without official data on fish captures. See also Table 5.

The negative relationship of waterbirds with fish catch is, to some extent, unexpected. Four out of five waterbird species increase in numbers along a period of fisheries' decline, even when only one is tipically non-piscivorous (Common Coot). However, it is possible that piscivores select different prey types or size classes than those commercially exploited (Liordos and Goutner, 2007). Besides eutrophication, the decrease in fish yield can be a consequence of overfishing, leading to a dominance of small fish due to the selective extraction of larger individuals (Van Rijn and Van Eerden, 2003). This larger biomass of small fish would be beneficial for species like Great Crested Grebe and Great Cormorant (Gwiazda, 1997; Smit *et al.*, 1997; Van Rijn and Van Eerden, 2003). Other species can survive entirely on prey other than fish, like Black-necked Grebe, a typical invertebrate-feeder (Jehl, 2001; Fjeldså, 2004).

Interaction among variables suggests that some biotic components might have effects on the relationship between waterbirds and trophic variables. Particularly jellyfish, the main top-down agent controlling the food web in the Mar Menor lagoon (Pérez Ruzafa *et al.*, 2002), seem to modify the response of waterbirds to nutrient enrichment, with a negative effect that suggests changes in food web structure (rather than simply a net effect on the quantity of resources). However, high collinearity among variables, reflecting parallel and interrelated dynamics at different scales and in various biotic compartments, call for a contentious interpretation of results. The complex nature of food-web interactions in eutrophicated waters (Vasas *et al.*, 2007) makes the interpretation of relationships and interactions difficult. The models have thus to be considered as simplifications helping to set out new hypothesis, to be tested by more detailed studies. It is also possible that the waterbird responses shown are site-specific, so comparative studies in similar ecosystems and in other habitat types are needed.

In the meanwhile, the relationships depicted by the models should help in the implementation of bird-based monitoring programs. The responsiveness of waterbird species and guilds to independent measures of wetland deterioration (nutrient loads, jellyfish blooms, fish declines) may be useful in the selection of metrics on which to build bird integrity indices (Bryce *et al.*, 2002; Van Strien *et al.*, 2009; Frederick *et al.*, 2009). Ecosystem managers should also take advantage from the signal value of some of these waterbird metrics. When eutrophication threatens coastal ecosystems, the early detection and management of nutrient inputs may be decisive in preventing a critical threshold from being reached, causing a dramatic mortality of macrophytes and a phase shift from a rooted macrophyte-dominated to a phytoplankton-dominated system. The greater vulnerability to eutrophication of nontidal areas, with long residence time of the water and prevailing sedimentation, makes the biological surveillance and advanced detection of this process a priority (Nienhuis, 1992). Classical dose–response relationships, the dose being nutrient load, the variable to be managed, and the response biomass or community composition of waterbirds, appear as optimal management tools (Philippart *et al.*, 2007; Krause-Jensen *et al.*, 2008), provided that enough financial support is given to the relatively easy task of monitoring bird metrics.

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Chapter 2. Structure and distribution of the waterbird community in the Mar Menor coastal lagoon (SE Spain) and relationships with environmental gradients

Introduction

Coastal lagoons are dynamic, wide open systems, and highly dependent on adjacent terrestrial and marine systems with which a continuous exchange of materials and energy occurs (Pérez Ruzafa and Marcos-Diego, 2003). A multitude of anthropogenic changes can modify the natural inputs from their watersheds (Menéndez and Comín, 2000; Herrera-Silveira *et al.*, 2002; Contreras and Warner, 2004). The confluence of opposing forces (terrestrial vs marine influence, natural processes vs anthropogenic modifying factors) help explain their heterogeneity, biodiversity and productivity (Tamisier and Bouduresque, 1994; Pérez Ruzafa *et al.*, 2002). Coastal lagoons and other transitional waters contribute considerably to global biodiversity, and are among the world´s most productive ecosystems (Van Den Bergh *et al.*, 2005). However, estuarine systems are also among the most threatened environments on the planet (Ma *et al*., 2007), suffering from harmful wastes and effluents, encroachment by urban development and overexploitation of resources (Essink, 2003; Álvarez-Rogel *et al.*, 2006; McKinney *et al*., 2006; Velasco *et al.*, 2006; García Pintado *et al.*, 2007). All these factors affect the biotic communities, by altering their patterns of distribution, abundance and use of resources (Ysebaert *et al.*, 2000; Pérez Ruzafa *et al.*, 2007). Monitoring these alterations can serve as a tool to assess the physicochemical and/or geomorphological changes occurring in transitional waters (Stolen *et al.*, 2004).

In the Mar Menor coastal lagoon (SE Spain), recent research has focused on tracking the influence of environmental degradation and, particularly, of eutrophication, on the long-term changes in composition and abundance of the bird community (Martínez *et al*., 2005; Robledano *et al*., 2008). The bioindicator role of waterbirds has been extensively reviewed and discussed (e.g. Peakall and Boyd, 1987; Adamus, 1996; Green and Figuerola, 2003; Gregory *et al*., 2003; Stolen *et al.,* 2004; Rönkä *et al.*, 2005) and is based on various avian properties (e.g. high trophic position, homeothermic character, high metabolic rate, complex behaviour). Also, waterbirds are suitable for low-cost monitoring through the use of citizen science (Kushlan, 1993).

Assessing the potential role of waterbirds as indicators of the ecological status of a coastal lagoon requires prior analysis of both the system status with respect to the reference (=undisturbed) ecological conditions and of the bird community. Waterbirds respond to the environmental variables that characterize the system, such as vegetation cover, salinity, pH, and trophic status; e.g. greater abundance and diversity of birds are expected in mesotrophic to moderately eutrophic wetlands. Although the variation of waterbird numbers can be related to changes in the supply of nutrients to coastal and inland waters (Nilsson, 1985; Raffaelli, 1999), their use as biological indicators has been questioned for the lack of a measurable direct response to changes in limnological variables (Adamus, 1996; Green and Figuerola, 2003). However, monitoring waterbird populations, at least in a regional system or a wetland complex (Adamus, 1996), could provide a good signal of environmental change (Martínez *et al.,* 2005), although one complication is that waterbird responses depend on the stage of eutrophication (Van Impe, 1985; Raffaelli, 1999; Van Eerden *et al.,* 2005), sometimes in a taxon-specific way (Rönkä *et al.,* 2005).

The status of waterbird populations has served as a criterion for establishing conservation priorities, as evidenced by the Ramsar list of wetlands of international importance. Therefore, in the Mar Menor (Ramsar Wetland since 1994) developing tools for monitoring and evaluating the waterbird community is a priority. Monitoring waterbirds in relation to environmental changes will help create a framework in which ornithological data can be linked to the quality of the ecological system. Based on previous studies searching for community responses to external processes (Martínez *et al.,* 2005; Robledano and Pagán 2006), this work seeks to describe the differential response of species to environmental gradients within the lagoon, as a way to increase understanding of their indicator value. Focusing on species which exclusively exploit the trophic resources of the water mass (shore habitats excluded), the objectives are to: [1] analyze the spatial and temporal distribution of the abundance and diversity of waterbirds during the annual cycle, [2] identify the major biological and environmental gradients that characterize the lagoon's internal variation, and [3] identify indicator species in relation to the spatial heterogeneity of the water mass, as reflected in meaningful zoning schemes.

Methods

Study area

The Mar Menor is a hypersaline coastal lagoon located in southeastern Spain (Fig. 1). With 135 km^2, it is the largest coastal lagoon of the western Mediterranean (Pérez Ruzafa and Marcos Diego, 2003) and has an average depth of 4 m. Almost enclosed from the Mediterranean sea by a sand bar (La Manga) with two mouths, it generally behaves as a sedimentary and concentrating basin, collecting water from several ephemeral channels.

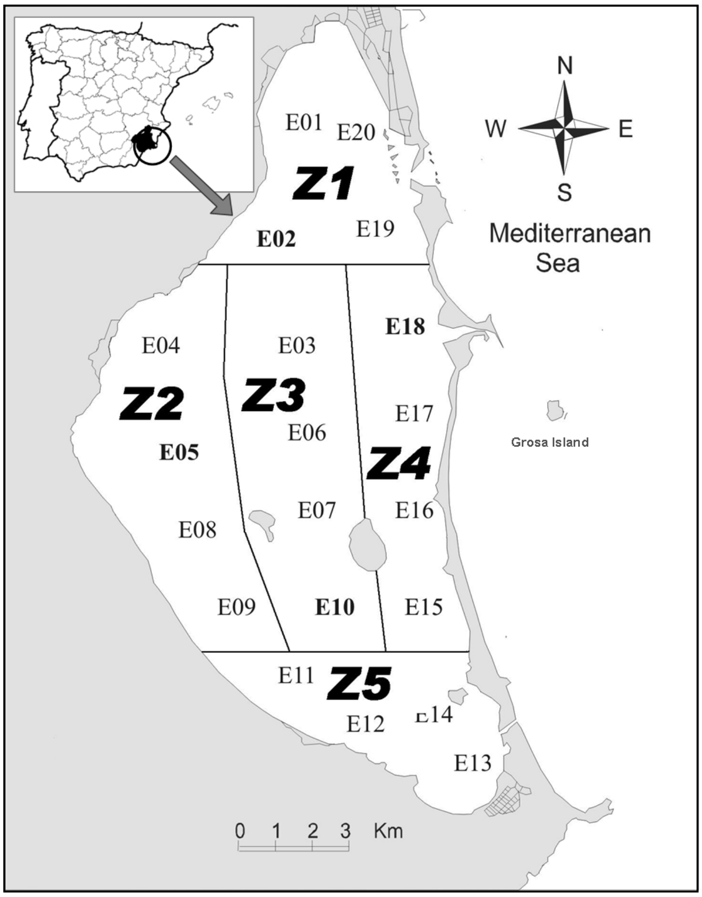

Figure 1. The Mar Menor lagoon: its geographical location in the Iberian Peninsula and Spanish administrative division. Location of the 20 census stations (“E”) within the 5 areas (“Z”) of the initial sampling scheme of the Mar Menor Lagoon is also shown.

The waterbird community is one of the most important biological components of the lagoon, and includes species of conservation concern for the European Union (Birds Directive 2009/147/EC, the codified version of Directive 79/409/EEC as amended) and species which serve as indicators of the effects of agricultural intensification and urban development (Hernández and Robledano, 1997, Martínez *et al.,* 2005).

Data collection

From October 2006 to March 2008 a monthly boat survey covered twenty sampling stations (“E” stations) distributed systematically inside the lagoon, a design proposed by Pérez Ruzafa *et al.* (2005) that follows a division in five zones (“Z” zones, Fig. 1) based on the degree of confinement (mostly related to marine influence, as defined by Guelorguet and Perthuisot, 1983) and productivity of the system (mostly associated with the input of nutrients from land). At each station all waterbird species were counted within a radius of 500 m around the boat. Basic meteorological conditions (wind, cloudiness and precipitation) were also recorded. Subsequently, the census was continued along a 500 m wide transect on either side of the boat, moving at a speed of 5 km/h for 7 minutes, giving a total sampling area for each station of 132.5 ha. The width of the strip was chosen on the basis of previous work in the area (Hernández and Robledano, 1997), 500 m being the threshold for ensuring the detection of smaller sized birds and flying species. Although the probability of detection within this band cannot be considered equal to 1, which might affect the consistency of the abundance estimates among sites and dates (MacKenzie and Kendall, 2002), the rule of proportionality would rarely have been violated (at least 95% of the individuals present were counted; Thompson, 2002), based on previous experience in the habitat and given its open and sheltered nature (a water table without obstacles). This was the reason to use a wider band than usually applied in strip-surveys of birds at sea (300-400 m), where the probability of detection may be lower even within this range (Barbraud and Thiebot, 2009). Double-observer surveys to adjust for detection probability (Nichols *et al.,* 2000) were not feasible (the capacity of the boat only allowed one observer), however, all the surveys were carried out by the same person to avoid observer bias. Three groups of environmental variables were used to characterize each sampling station (Table 1): [1] descriptors of average physicochemical and trophic conditions of the water, taken from Dirección General de Costas and Confederación Hidrográfica del Segura (2007), [2] directly recorded climate variables, and [3] variables expressing distances to shelter or disturbance elements (e.g. islands and airports, respectively), measured with a Geographic Information System (GRASS 6.3).

Name	Definition	Source
DEPTH	Monthly mean depth of sampling station	*In situ*
STRESS	Mean weighted record of disturbance events during the census in each sampling station	*In situ*
WINDS	Mean intensity of wind at each sampling station	*In situ*
CLOUDS	Mean cloud cover at each sampling station	*In situ*
FNETS	Mean frequency of fishing nets at each sampling station	*In situ*
SALIN	Salinity (g/l)	D.G.C.C.H., 2007
TURB	Turbidity (u.r.)	D.G.C.C.H., 2007
OXDIS	Dissolved oxygen in water (mg/l)	D.G.C.C.H., 2007
CLORT	Total chlorophyll (mg/m^3)	D.G.C.C.H., 2007
DGOLA	Distance to main communication channel (m)	GRASS 6.3
DHARB	Distance to the nearest harbour (m)	GRASS 6.3
DSHORE	Distance to the nearest shore (m)	GRASS 6.3
DISLE	Distance to the nearest island (m)	GRASS 6.3
DALB	Distance to the main discharge channel into the lagoon (m)	GRASS 6.3
DAIR	Distance to airport (m)	GRASS 6.3
DURB	Distance to the nearest village (m)	GRASS 6.3

Table 1. Codification, description and sources of the environmental variables used to characterize sampling stations and included in multivariate analyses. D.G.C.C.H., 2007 = Dirección General de Costas and Confederación Hidrográfica del Segura (2007)

Statistical analyses

Census results were averaged by Zone (Z) and season (only in the case of winter, since there are two wintering seasons). To avoid the influence of random contacts with occasional species, those appearing only once were dropped from the analysis, following Ysebaert *et al*. (2000). Abundance data were checked for normality and homogeneity of variances, and non-parametric statistics (Kruskal-Wallis Test) were used when these were not achieved after data transformation. Alpha diversity and numeric diversity (or k-dominance) were calculated from the raw values of abundance, separately for winter (October-March) and summer (April-September). Unlike alpha diversity, which includes the relative proportion of all species (from the most abundant to the rarest), k-dominance expresses the diversity in terms of the most abundant species (very few species dominant, high value). Unless otherwise stated, statistical analyses were performed with the R software package (R development Core Team, 2007). Monthly abundances for “Z” zones were obtained by summing the species counts for the four “E” stations that made up each “Z” zone. To represent individual species’ seasonal abundances, the mean of the monthly abundances was calculated for summer and winter periods. A more detailed examination of seasonal changes was made (only in the case of total abundance) among shorter periods (spring, March-May; summer, June-August; autumn, September-November; winter, December-February). Also, to represent the numerical importance of each guild (distinguishing grebes, cormorants, gulls and terns), their percent contribution to total abundance was calculated for each month of the annual cycle.

The relationship between spatial and temporal variations in the waterbird community and the environmental heterogeneity of the lagoon was studied through multivariate analysis using the Primer 6 software. Multivariate methods of ordination and classification are often used to describe the degree of similarity (or dissimilarity) between areas and seasons in terms of species composition (Zárate-Ovando *et al.*, 2008). The analysis in this study was restricted to the wintering community, which, in global terms, is more abundant and richer than the summer assemblage. Further, as shown by previous studies (Hernández and Robledano, 1997; Martínez *et al.*, 2005; Robledano *et al.,* 2008), the wintering community is expected to be more responsive to spatial changes in the trophic status of the water. Data were normalized using log (x +1) transformation before a Multi-Dimensional Scaling (MDS) ordination was carried out to analyze the degree of similarity in waterbird community structure between sampling stations. This multivariate method allows sampling units to be classified on the basis of similarity levels, and to be labelled according to different factors or characteristics of interest. The first factor used was “basin” (north/south), and then the original “Z” zones. Since these zones did not correspond to specific waterbird assemblages, a new factor (“waterbird zoning”) was established based on the level of similarity which had a more coherent spatial expression (80-85% similarity). Subsequently, SIMPER analyses were performed to identify which species were indicative of differences between groupings, based on the two factors of spatial classification: “basin” and “waterbird zoning”.

Kruskall-Wallis tests were used to assess the statistical significance of differences in structural parameters between these groupings.

Following a widely used approach to the description of change in waterbird community structure along environmental gradients (Ysebaert *et al.*, 2000; Van Den Berg *et al.*, 2005; Ma *et al.*, 2007; Mander *et al.*, 2007), a Principal Component Analysis (PCA) was conducted after data normalization, in order to detect redundancy between environmental variables (Coreau and Martin, 2007) and to reduce them to a few major components that described the main gradients. Finally, data were submitted to Canonical Correspondence Analysis (CCA), which is a useful technique for relating community composition with spatial and temporal variations of known environmental variables (Ter Braak, 1986). Environmental variables were also log (x +1) transformed to eliminate the effect of different scales of measurement and to reduce the influence of very large values. The CCA ordination diagram displays the distribution of species along each of the environmental variables (vectors) and the relationship between variables and species based on differences between sampling areas (Poulin *et al.*, 2002; Bolduc and Afton, 2004).

Results

Spatio-temporal variation in alpha diversity and k-dominance

The results of the censuses are shown in Table 2. The lagoon as a whole underwent a significant temporal decrease in alpha diversity (Kruskal-Wallis chi-squared = 19.93, df = 1, p = 7.992e-06) from winter, when the southeastern zones (Z4 and Z5) showed the highest values, to summer, when the highest mean values shifted to the central and northern zones (Z3 and Z1) (Figure 2). Alpha diversity increased during the months of transition between these two main seasons, particularly in the central zone (Z2 and Z3; Fig. 2). In global terms alpha diversity values did not differ significantly among zones or latitudinally within the lagoon (Kruskal-Wallis test, p=0.94 and 0.97), but did so between the more detailed seasonal periods, with autumn and winter showing the highest values (Kruskal-Wallis chi-squared = 19.46, df = 3, p = 0.0002).

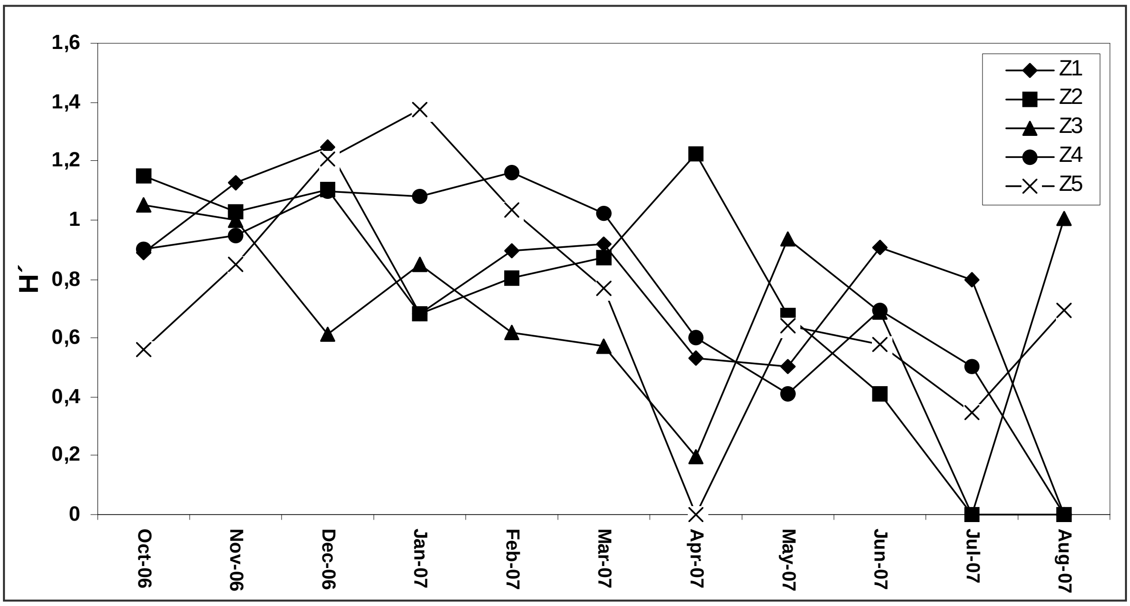

Figure 2. Variation in Shannon-Wiener diversity in each "Z" area during a complete annual cycle (2006-2007).

The k-dominance index in winter was fairly similar between areas, needing 4 to 5 species to accumulate 100% of the total abundance observed. The southwestern section, which includes the most confined areas, Z2 and Z5, showed slightly lower dominance values than the rest of the lagoon, that is, the contribution to total abundance was divided among more species (Figure 3). In summer in all areas (Fig. 3), the first and second species made a greater contribution to abundance than in winter. However, the lower slopes of the curves point to a more even contribution on the part of abundant species. The most open area (Z4) showed the lowest dominance in summer, with five species needed to accumulate over 90% of the total abundance recorded (vs only three in winter), followed by the central zone (Z3). In contrast, the more confined areas (Z2 and Z5) showed the greatest dominance in summer. In winter, the southern zone (Z5) showed the highest alpha diversity and the lowest k-dominance, with a more balanced weight between species than in summer.

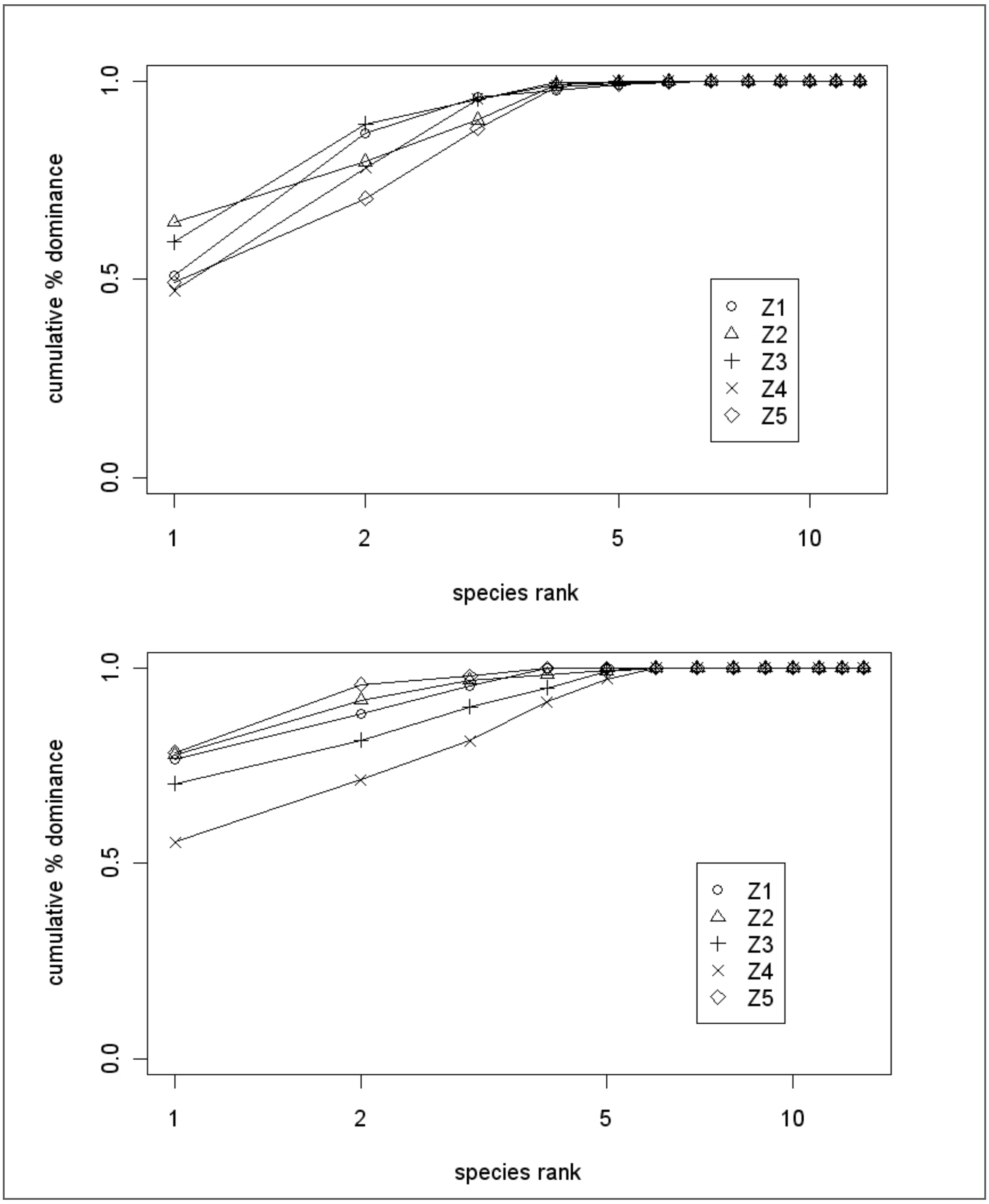

Figure 3. K-dominance curves for each “Z” area calculated from the seasonal average (upper graph, winter; lower graph, summer).

Looking at both indexes toghether, the southeastern section (Z5 and Z4 zones) showed the highest overall diversity in winter, and the northeastern one (Z1 and Z4) in summer. The central, more homogeneous zone (Z3), held an intermediate position in both seasons. However, based only on winter alpha diversity values, the differences among the five zones were not significant (Kruskal-Wallis chi-squared = 4.44, df = 4, p = 0.34).

	Winter (averaged 2006/07 and 2007/08)					**Summer (2007)**				
	Z1	**Z2**	**Z3**	**Z4**	**Z5**	**Z1**	**Z2**	**Z3**	**Z4**	**Z5**
Podiceps cristatus (POCR)	3.13	5.92	7.50	2.35	8.25	0.00	1.60	0.00	0.40	0.00
Podiceps nigricollis (PONI)	21.58	32.75	29.25	30.08	32.67	0.00	1.00	0.60	0.00	0.00
Phalacrocorax carbo (PHCA)	4.05	5.25	3.83	17.27	31.00	0.00	0.00	1.00	0.20	0.00
Larus michahellis (LACA)	14.78	19.58	7.67	11.19	15.58	8.40	9.80	10.4	2.80	6.80
Larus ridibundus (LARI)	0.83	0.33	0.00	2.58	0.67	0.60	0.00	0.20	0.00	0.20
Larus genei (LAGE)	0.08	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Larus audouinii (LAAU)	0.00	0.08	0.00	0.00	0.08	0.00	0.00	0.00	0.20	0.00
Sterna sandvicensis (STSA)	0.10	0.00	0.08	0.00	0.00	0.60	0.20	0.00	0.80	0.00
Sterna hirundo (STHI)	0.08	0.00	0.00	0.00	0.00	2.40	0.40	0.60	0.20	0.20
Mergus serrator (MESE)	0.00	0.00	0.17	0.83	0.50	0.00	0.00	0.00	0.00	0.00
Egretta garzzetta (EGGA)	0.00	0.00	0.00	0.00	0.00	0.00	0.20	0.60	0.00	1.20
Total abundance	44.65	64.00	48.50	64.31	88.75	12	13.2	13.4	4.60	8.40

Table 2. Census results grouped according to the zoning of Pérez Ruzafa *et al.* (2005). Values shown are seasonal mean abundances of each species at the four “E” sampling stations belonging to each “Z” zone (winter, October to March; summer, April to September).

Spatio-temporal variation in abundance and guild structure

Abundance was greater in autumn and winter than in spring and summer (Kruskal-Wallis chi-squared = 100.02, df = 1, p-value < 2.2e-16; Figure 4), the southernmost zone (Z5) displaying the highest abundance in winter (mean = 77.7 individuals), although, as occurred with diversity, the values decreased in summer. The innermost zone (Z2) also presented a winter peak of abundance (mean = 73.7), while summer maxima shifted to areas further away from La Manga (Z1, Z2 and Z3), possibly due to increased disturbance in this area.

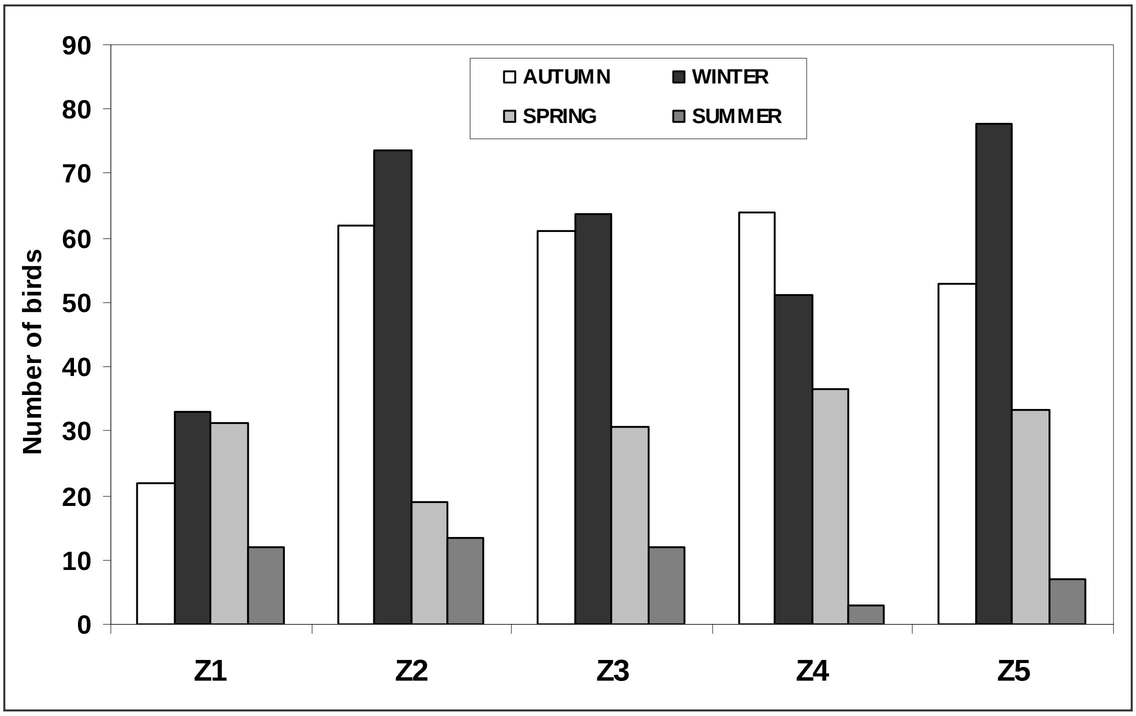

Figure 4. Spatio-temporal variation of total abundance in each Z area between seasons during the annual cycle 2006-2007 (values averaged from monthly abundances).

To assess their contribution to total abundance, species were grouped into guilds that were differentiated according to their taxonomic origin and feeding strategy: grebes, terns, gulls and cormorants (analysis restricted to the 2006-07 annual cycle). In winter the lagoon was dominated by grebes (62.8%), followed by gulls (22%) and cormorants (15.1%). The contribution of wintering terns, due exclusively to the Common Tern (*Sterna hirundo*), is negligible (<1%). In summer, the dominant community consists of gulls (80.1%) and terns (9.3%).

Identification of indicator species

All the sampling stations (except E18 and E20) showed more than 70% similarity, meaning that, during winter, inter-station differences are slight and a high degree of overlap exists in the spatial distribution of waterbirds. However, the MDS ordination of the cases, labeled by the factor "basin", showed differences in community composition between the two main basins (stress = 0.14). Taking the lagoon’s zonation as an expression of its internal heterogeneity, this same ordination procedure, but classifying the cases according to the “Z” zoning, showed that changes in community structure poorly reflected the zonation established by Pérez-Ruzafa *et al.* (2005). A new spatial aggregation of cases was derived from the “waterbird zoning” classification (Figure 5) in which E18 and E20 sampling stations were grouped under one category, giving greater weight to their dissimilarity from the rest of the lagoon than to their similarity.

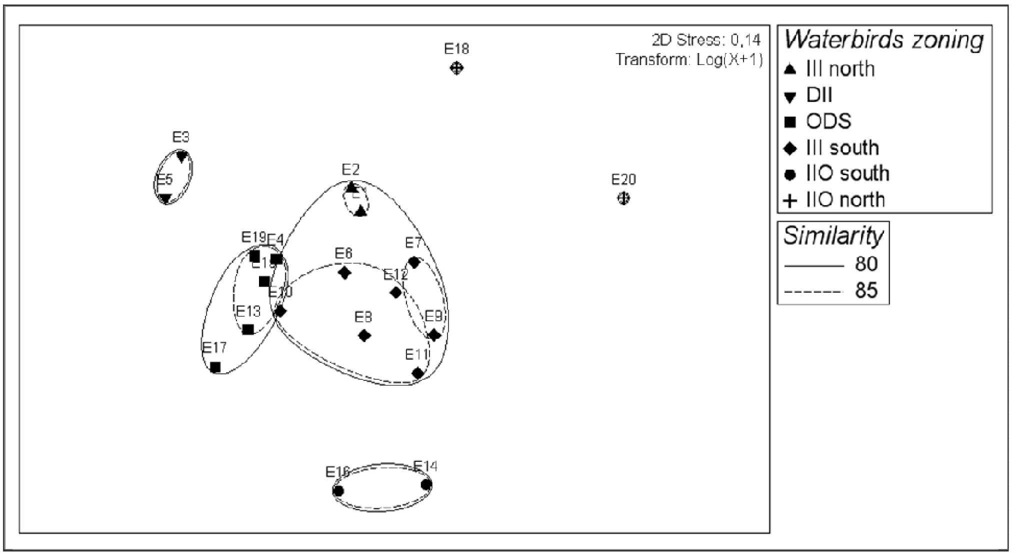

Figure 5. Two-dimensional MDS ordination diagram of sampling stations (winter cases) labelled according to the new zoning based on waterbirds. The 80-85% similarity level was taken for delimiting groups which have a coherent spatial expression. The original resemblance matrix based on Bray-Curtis similarity has been log (x+1) transformed.

The new zones (spatial projection on the lagoon in Figure 6) can be characterized from a combination of terrestrial influence (direct or indirect, depending on their relative position regarding incoming channels and internal circulation), marine influence (inner or outer, in relation to the open sea) and latitudinal position (north or south): IIIN (indirect influence inner north), DII (direct influence inner), IIIS (indirect influence inner south), IIOS (indirect influence outer south), IION (indirect influence outer north) and ODS (other dispersed sampling stations). “Indirect influence” refers to the effect of nutrient inputs through internal circulation.

The results of SIMPER analysis for the factor "basin" (Table 3) indicated that the Great Cormorant was the species that contributed most to the differences between basins (24.41% dissimilarity), followed closely by the Yellow-legged Gull (*Larus michaellis*) and the Great-creasted Grebe (*Podiceps cristatus*). Black-headed Gull (*Larus ridibundus*) was the main representative of the north basin (mean abundance four times higher).

BASIN			
North & South Basin	Average dissimilarity= 26.32		
	Average Abundance North	Av.Abund South	Contribution %
PHCA	0.71	1.30	24.41
LACA	1.21	1.64	22.99
POCR	0.51	1.10	22.34
PONI	1.97	2.38	16.29
LARI	0.19	0.05	6.43
WATERBIRD ZONING			
DII & III$_S$	Av. diss=31.39		
	Av.Abund DII	Av.Abund III$_S$	Contrib %
LACA	0.56	1.80	40.11
POCR	0.37	1.22	27.71
III$_N$ & IIO$_S$	Av. diss=30.52		
	Av.Abund III$_N$	Av.Abund IIO$_S$	Contrib %
PHCA	0.62	3.15	63.10
POCR	0.45	0.88	12.27
MESE	0.00	0.17	4.46
DII & IIO$_S$	Av. diss=40.59		
	Av.Abund DII	Av.Abund IIO$_S$	Contrib %
PHCA	0.66	3.15	52.90
LACA	0.56	1.71	24.17
POCR	0.37	0.88	10.20
III$_S$ & IIO$_S$	Av. diss=26.17		
	Av.Abund III$_S$	Av.Abund IIO$_S$	Contrib %
PHCA	0.84	3.15	61.15
POCR	1.22	0.88	13.57
DII & IIO$_N$	Av. diss=38.98		
	Av.Abund DII	Av.Abund IIO$_N$	Contrib %
LACA	0.56	1.80	39.24
LARI	0.00	0.67	19.05
IIO$_S$ & IIO$_N$	Av. diss=40.77		
	Av.Abund IIO$_S$	Av.Abund IIO$_N$	Contrib %
PHCA	3.15	0.73	45.66
POCR	0.88	0.06	15.08
LARI	0.00	0.67	11.96
LACA	1.71	1.80	6.36

Table 3. Results of SIMPER analysis (Similarity Percentages), which identifies the indicator species contributing most to differences in winter mean abundance between the north and south basin (factor “basin”) and between zones derived from MDS ordination (factor “waterbird zoning”). POCR=*Podiceps cristatus*; PHCA=*Phalacrocorax carbo*; MESE=*Mergus serrator*; LARI=*Larus ridibundus*; LACA=*Larus michaellis* (formerly *Larus cachinnans*)

In the SIMPER analysis of the classification factor "waterbirds zoning" (Table 3) the greatest differences were found between the two outer areas IIO_N and IIO_S, with Great Cormorant making a 45.66% contribution to these differences (due to its preference for IIO_S). When comparing this area with those characterized by different combinations of terrestrial influence and latitudinal position (III_N, DII and III_S), the Great Cormorant always explained most of the difference (50 to 60%), and was the most characteristic species of the southern part of the external shore. The restricted presence of Red-breasted Merganser in IIO_S should also be mentioned. Black-headed Gull was the main representative of IIO_N, the highest difference for this species being established between IIO_N and DII. A comparison of areas with varying degrees of terrestrial influence (increasing from III_S to DII), showed that the main differences were due to Yellow-legged Gull (40.11% contribution) and Great-crested Grebe (27.71%). Moreover, the Great-creasted Grebe showed a preference for IIO_S rather than III_N (south *vs* north Basin) and for III_S rather than IIO_S (internal *vs* external shore). Black-necked Grebe showed a fairly homogeneous distribution, as reflected by its general contribution to internal similarity in all areas, except IIO_S (characterized by the abundance of Great Cormorant) and III_N (characterized by the abundance of Yellow-legged Gull). This last species is widely distributed throughout the lagoon with a generally high, but locally changing, abundance, making it one of the species that often contributes to differences between areas. However, taken together, the two *Larus* species showed a preference for outer areas that are well connected with the Mediterranean Sea (IIO_N and IIO_S).

When the new zones were compared on the basis of total waterbird and guild abundances, the former showed significant differences in winter (Kruskal-Wallis chi-squared = 27.98, df = 5, $p<0.001$), but not in summer. In winter, grebes, cormorants and gulls showed significant differences among zones (Kruskal-Wallis chi-squared = 15.12, 32.98, and 24.69, respectively; df = 5, $p<0.001$), particularly cormorants (favouring IIO_S over other zones), while grebes showed higher abundance in IIIs, but only compared with IIO_N. Gulls were widely distributed in the lagoon, favouring III_N and avoiding DII. In summer, gull abundance varied significantly between zones (Kruskal-Wallis chi-squared = 14.49, df = 5, $p<0.05$), favouring III_N and IIO_S (closer to saltpans and to the Mediterranean) as did tern abundance in III_N and IIO_N (northernmost areas open to Mediterranean influence).

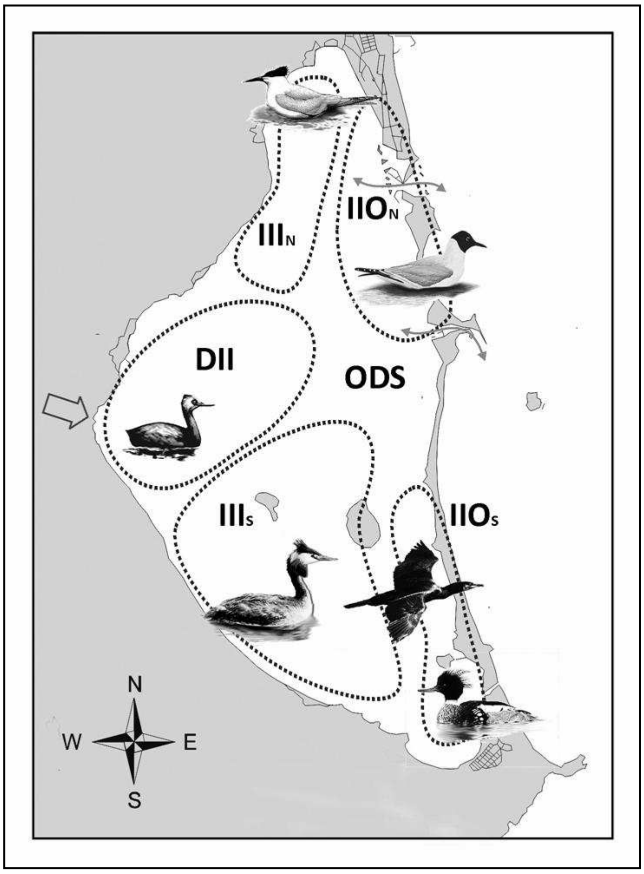

Figure 6. Spatial projection of the new zoning based on the relationship of the waterbird community with the degree of confinement and terrestrial influence. Zones are identified by their most representative species

Identification of gradients

The first axis of the Principal Component Analysis, which explained 32.2% of variation (Table 4), pointed to a gradient of confinement that depended on the location of the stations with respect to the channels (mouths) of communication with the Mediterranean Sea (DGOLA) and the outlet of the main channel which empties into the lagoon (DALB). Confinement is considered here in a purely geomorphological sense, and therefore represents an "internal state gradient". Stations of the ODS group of MDS ordinations appear mostly on the negative side of the gradient reflecting their closeness to the mouths and their distance (and sheltered position) from incoming watercourses. The second axis, accounting for 22.6% of the explained variation, described a gradient of disturbance, in which the extremes were defined by the distance to "shelter elements" (DISLE on the positive end, the likelihood of birds being disturbed increasing with distance from islands), and to potentially stressful or disruptive elements, namely airports (DAIR, on the negative end). The STRESS variable, generated on the basis of the mean weighted record of disturbance events during the census (transit of boats, water sports, etc.), was also associated with the positive side of the gradient. The third axis (16.6% of explained variation) seemed to represent a gradient which roughly described ecosystem productivity, expressed through variables related with photosynthetic activity (TCLOR and OXDIS) on the positive side, and with STRESS on the negative side (unproductive areas are also more disturbed by human presence). Consequently, areas of the southwestern shore (DII and III_S) which still maintain a significant degree of natural conditions (low land-based human disturbance), and which experience a general increase in productivity, were located at the positive end of this gradient.

	PC1	**PC2**	**PC3**	**PC4**	**PC5**
Eigenvalues	5.16	3.62	2.66	1.23	1.09
% Cumulative Variation	32.2	54.9	71.5	79.2	86
Eigenvectors					
DEPTH	0.189	-0.254	-0.358	0.209	-0.13
WIND	-0.368	-0.184	-0.063	-0.3	-0.021
SALIN	-0.112	-0.394	-0.148	0.204	-0.015
TURB	0.263	-0.071	0.358	-0.31	-0.04
OXDIS	0.243	-0.29	0.312	-0.109	0.027
CLORT	0.102	-0.19	0.462	0.151	-0.201
DGOLA	0.326	0.117	0.293	0.049	-0.149
DSHORE	0.342	-0.063	-0.328	-0.169	0.171
DISLE	-0.107	0.465	0.124	-0.095	0.214
DALB	-0.37	-0.045	-0.041	-0.125	0.285
DAIR	-0.065	-0.495	0.026	0.126	0.104
DURB	0.335	-0.007	-0.266	-0.268	0.276

Table 4. Eigenvalues, explained variation for principal components generated in PCA, and coefficients of environmental variables included in the analysis.

In the Canonical Correspondence Analysis which jointly displays the community structure and the main environmental features of the lagoon (Figure 7), the ordination was significant at $p<0.01$ (function anova.cca; R package “vegan”), but explained a low proportion of variance (25.5% in total, with 18% between the first two axes) and therefore must be interpreted with caution. However, the first axis seemed to differentiate species according to their foraging strategy and tolerance to anthropogenic activities or elements. At the positive end, Black-headed Gull, Slender-billed Gull (*Larus genei*), Sandwich Tern and Common Tern were associated with areas far from the islands and from the shores of highest primary productivity (i.e. areas with low concentrations of total chlorophyll), and also with areas close to harbours and which have a small number of fishing nets. Audouin´s Gull (*Larus audouini*), Yellow-legged Gull, Great-creasted Grebe and Black-necked Grebe were found around the centre of coordinates, whereas Red-breasted Merganser and Great Cormorant were slightly displaced towards the negative end, because of the greater preference of the two latter species for food-rich areas characterized by the presence of fishing nets and shelter elements (islands).

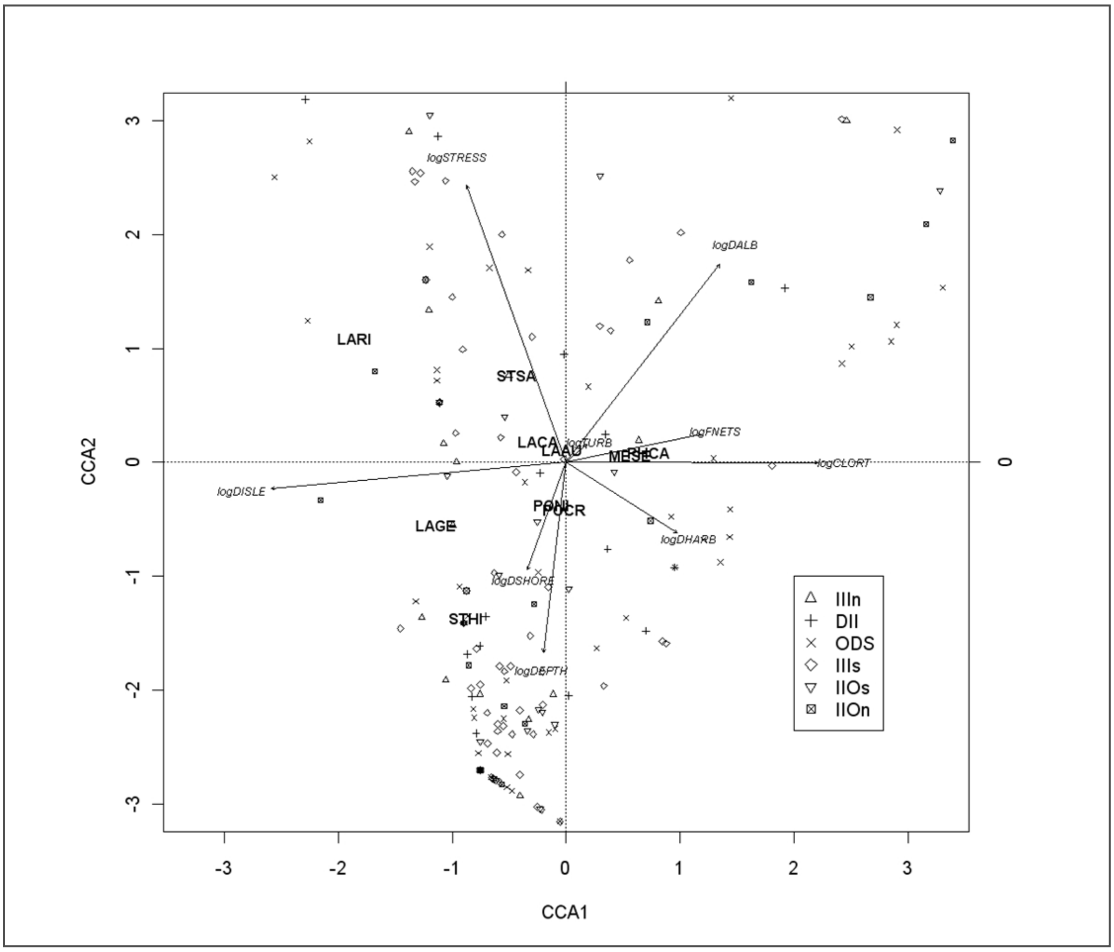

Figure 7. Canonical Correspondence Analysis biplot of wintering waterbird species (average of 2006-2007 and 2007-2008 winter periods) and environmental variables. Species abbreviations as in Table 2. Cases are labelled according to the new “Z” zoning based on waterbirds.

The second axis was more difficult to interpret since it differentiated stations according to their intrinsic physical characteristics, with deep areas far from the urban shores at the negative end, and shallower areas of the north basin associated with the STRESS variable at the positive end (Fig. 7). Following this interpretation, Black-headed Gull and Common Tern, located in the same position with respect to axis I, were clearly separated on axis II (the first species is related with IION and the second with DII and IIIS). In a similar way, gulls (except *Larus genei*) tended to be separated from species that favour deeper sections, due to their capacity to dive (grebes) or to use other foraging techniques suitable for these areas.

Discussion

The use of indicators based on bird numbers has been critiziced (Morrison, 1986; Landres *et al*., 1988). However, provided that the attributes that the birds are supposed to indicate (e.g. hydrology, nutrient status, contaminants, human disturbance) are clearly defined (Stolen *et al*., 2004), avian monitoring programs can be used to construct policy-relevant indicators (Van Roomen *et al*., 2006). Although waterbird censuses are the most common form of monitoring (Hoffmann *et al*., 1996), they have not been widely used to assess the ecological integrity of Mediterranean wetlands (but see e.g. Green *et al*., 2002; Paracuellos *et al*., 2002; López *et al*., 2010).

The waterbird community of the Mar Menor seems to respond to internal functional gradients, as well as to structural features of the lagoon and to characteristics of the surrounding habitats. Both internal and external factors related with waterbird abundance and distribution are affected by human activity, as has been shown in other wetland ecosystems, both at local and landscape scales (Kerekes and Pollard, 1994; Noordhuis *et al*., 2002; Paracuellos 2006; Paszkowski and Tonn, 2006; Hebert *et al*., 2009).

Lagoons are characterised by both among and within-ecosystem heterogeneity (Basset *et al*. 2008). Spatial and temporal changes in waterbird abundance may be also dramatic, particularly between seasons (Zárate-Ovando *et al*., 2008), but not necessarily in parallel to changes in other biological components (Maci and Basset, 2009). In the

Mar Menor there is a general drop in diversity and abundance from winter to summer, a characteristic of southern coastal wetlands which receive an influx of wintering waterbirds, becoming key refuges for some species (Scott and Rose, 1996; Erwin, 1996). Within the lagoon, there is also a shift in peak diversity and abundance values from the south to the north basin in summer, particularly to areas closer to the open sea. These northern areas generally show the lowest variability in abundance and diversity, which may be explained by a constant influence of human activities (boat traffic, fishing), limiting its use by waterbirds. The proximity of a Regional Park (Salinas y Arenales de San Pedro del Pinatar) can partially offset this limitation by offering an alternative refuge and food-rich habitat (saltpans). The species found closer to the Mediterranean Sea also benefit from the presence of the natural mouth of the lagoon (Las Encañizadas), a mosaic of mudflats, shallow channels and halophytic scrub. Such landscapes may act as a key factor in the birds' choice of habitats in which to settle (Taft and Haig, 2006).

Constant disturbance and low productivity in the north basin, contrasts with the fact that some species are concentrated there, like the Sandwich Tern or the Black-headed Gull. Such preferences may be explained by species-specific sensitivity to disturbance (Blumstein *et al*., 2003; Burton, 2007), by the choice of a specific foraging habitat, or by the availability of preferred prey, rather than resource abundance per se (Fasola *et al*., 1989). Also, foraging strategies may have some influence, especially in the case of terns, whose aerial hunting technique could better tolerate human presence. The drop in diversity and abundance in the south basin in summer may respond to the marked increase in human disturbance that occurs at this time. Nutrient inputs from agricultural sources seem also to influence the patterns of abundance of the three main wintering guilds (grebes, gulls and cormorants; Fig. 6). During winter, nutrient inputs into the south basin may be leading waters towards more eutrophic conditions, especially in confined areas. In the most directly affected areas (like Z2) this phenomenon would explain the presence of a few species but in high abundance (DeLuca *et al*., 2008). In the meanwhile, the area of indirect influence (Z5) would experience only a moderate trophic enrichment, favouring the presence of more species with more balanced abundance.

While at community level winter abundance and diversity seem to respond to gross levels of productivity, individual species' responses are also indicative of internal gradients. The Great-creasted Grebe, characterized by its early response and an intermediate tolerance to eutrophication (Fjeldså, 2004; Esteve *et al.*, 2008), favours areas indirectly influenced by nutrient inputs (Z5) over direct discharge zones (Z2). In view of such preferences the Great-crested Grebe seems to represent an early warning of moderate levels of eutrophication (already recognized as such in the 1980s; see Hernández and Robledano, 1997). The same preference is shown by the Great Cormorant, which has been proposed as an indicator of waters of intermediate turbidity (Van Rijn and Van Eerden, 2003; Rönkä *et al.*, 2005). However, in the study area the Great Cormorant should preferably be considered a background indicator of eutrophication, given its mobility and landscape-scale response to increased productivity and complementary food sources (irrigation ponds, fish farms). Compared with earlier studies (Hernández and Robledano, 1997), the main difference revealed by our results, apart from the decrease of Anatidae, is precisely the increase of cormorants in the winter waterbird assemblage.

Of the major guilds, diving piscivores favour the confined, nutrient enriched waters of the southern half of the lagoon, although the preferences of grebes are not so clear because of the wide-ranging distribution of Black-necked Grebe. This species can feed at most water depths and is able to exploit opportunistically various sources of short-lived, superabundant food (Fjeldså, 2004). It was the most abundant species (Table 2), found throughout most of the lagoon, including areas directly affected by nutrient inputs. Gulls and terns, usually feeding on the surface or from the air, concentrate in more oligotrophic waters near saltpans and other open areas close to the lagoon's mouth.

In short, the results highlight the existence of a diversity gradient whose extremes are associated with the opposing land and sea influences on the lagoon (external factors). Diversity increases with proximity to the Mediterranean Sea, which can be attributed to the increased diversity of food resources and feeding habitats of the transitional area between the lagoon and the sea (Pérez Ruzafa *et al.*, 2007). Superimposed on this gradient, there is a latitudinal gradient of bird abundance (increasing southward in winter and vice versa in summer), which seems linked to changes in the level of human disturbance within the lagoon (internal factor). Therefore, an anthropogenic modifying factor, which is very relevant for birds, must be added to the original zoning based on functional gradients of the lagoon (Pérez Ruzafa *et al.*, 2005). In contrast to other biota of transitional water ecosystems, the responses of birds seem to integrate the combined effects of purely physical and/or trophic influences on the one hand, and the human deterioration of habitats directly or through landscape modification (Rosa *et al.*, 2003; McKinney *et al*., 2006; DeLuca *et al.*, 2008, but see also Traut and Hostetler, 2004). Since human pressure on the lagoon at different scales (watershed, immediate landscape, shoreline, lagoon sections) occurs through relatively discrete and spatially well-referenced paths, the distribution of waterbirds exploiting the water mass emerges as a useful barometer of the system's response.

The physical structure and geomorphology of the lagoon can help to explain the differential use of the water mass by some species. Great Cormorants, for example, feed mainly in the area with more islands (IIOS), where they can perch on small emergent rocky outcrops (Reymond and Zuchuat, 1995; Roycroft *et al.*, 2007). Also, restricted to the island-rich part of the lagoon is the Red-breasted Merganser, once a characteristic species of the lagoon but now undergoing a steady decline in numbers (Martínez *et al.*, 2005). The positive numerical response of other piscivores (cormorants and grebes) in some phases of eutrophication (Esteve *et al.*, 2008) suggests that changes in the type and quality of food resources might explain the Merganser's opposing trend.

The species favouring the northern, island-free part of the lagoon, seem to be more dependent on the influence of the Mediterranean Sea than on increased productivity of terrestrial origin. An example is the Black-headed Gull, which shows a strict preference for the more open and oligotrophic waters of the north basin. Black-headed gulls have increased notably in the Mediterranean during the last century in response to the increased variety and quantity of resources offered by man, although this process has occurred at a landscape level (Blondel and Aronson, 1999). In the Mar Menor, the zones favoured by Black-headed gulls do also display the greatest diversity of trophic resources and are the most suitable for the species' foraging techniques (Lewis *et al*., 2003).

The final ordination analysis (CCA) reinforces the idea that the numerical and distributional responses of wintering waterbirds to internal gradients are modified by the structure of the surrounding landscape, the geomorphology of the basin, and the extent of human disturbance. It shows how the feeding strategies and feeding distribution patterns of the species are broadly related to particular features or variables. Only Great Cormorant and Red-breasted Merganser associate with surrogates for primary or secondary productivity (chlorophyll and fish nets), as well as with variables that represent confinement and shelter. In other Mediterranean lagoons the piscivorous guild does also benefit from fishery management (Crivelli *et al.*, 2005; Boldreghini and Dall'Alpi, 2004). Given the increasing numbers of cormorants wintering in the Mar Menor lagoon (Martínez *et al.*, 2005; Esteve *et al.*, 2008), managers should be aware of the potential conflicts with local fisheries (Engström, 2001). Gulls and grebes, which look less dependent on physical elements, fishing structures or sheltered areas, segregate spatially in the lagoon according to foraging preferences or sensitivity to anthropogenic stress.

Taken together, the results show the suitability of incorporating waterbirds in a monitoring scheme of the lagoon, where they act as integrative indicators of the spatial trends of impairment (Zárate-Ovando *et al.*, 2008; Hubina, 2008). Despite being relatively easy to perform, bird monitoring schemes tailored to the internal heterogeneity of coastal lagoons are rarely used in the Mediterranean. (for exceptions see, e.g. Boldreghini and Dall'Alpi, 2008; Hubina, 2008). Comparative studies among similar sites undergoing different stages of alteration would be especially valuable. The interpretation of the ecological preferences and responses of waterbirds to environmental stress may have a variety of additional applications for conservation. Firstly, they can serve to assess the conservation status of species at local and regional scales, and, secondly, they could serve as a warning signal about the changes that might occur in similar ecosystems in earlier stages of deterioration.

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Chapter 3. Lagoons as waterbird habitat: response of communities to human impact and management across space and time scales

Introduction

Mediterranean lagoons: values and impacts

Coastal lagoons (CLs hereafter) are transitional waterbodies which harbor remarkable biological values and resources while performing key functional roles for ecological and social systems (Pérez-Ruzafa and Marcos, 2012). Depending on the bioclimatic and geographical setting, physical dynamics, historical trends of land reclamation and the stage of ecological succession in which they are found, CLs range along a wide gradient of size, depth, terrestrial influence, connectedness to the sea and internal heterogeneity (Pérez-Ruzafa *et al*., 2011). They often appear as coastal wetland complexes in which several types of waterbodies and vegetated wetlands coexist. Human uses like fishing, hunting, navigation, fish rearing and salt harvesting have historically shaped the configuration of such wetland complexes, and are still conditioning their ecological functioning today. In the Mediterranean, most CLs are managed for fishing or fish culture, a type of resource use that can be developed with minor morphological changes (although often with critical effects on hydrological functioning). Surrounding land uses like agriculture and tourism have also historically modified the landscape setting of coastal plains and shorelines, becoming much more widespread and intensive in recent decades. Even in preserved CL systems (Nature reserves and Ramsar Sites), the indirect influence of such intensification results in water eutrophication and general loss of biological value (Aliaume *et al*., 2007).

Of particular landscape impact has been, among the traditional wetland uses, the transformation of shallow basins and wetland sections for salt exploitation. Hyperhaline aquatic environments like exploited solar saltworks, salterns or salinas (Evangelopoulos *et al*., 2008) can be important alternative habitats for waterbirds and other biota (Oliva-Paterna *et al*., 2006; Amat *et al*., 2007; Ma *et al*., 2010). The abandonment of these artificial wetlands results in a dramatic change in land and water use, with a high risk of losing values of economic, cultural and environmental importance (Paracuellos *et al*., 2002; Crisman *et al*., 2009), including their potential use for the treatment of nutrient-enriched waters (Crisman, 1999; Marques *et al*., 2003). Ecotourism and other activities, coupled with salt production, are also compatible with the maintenance of most of the cultural, landscape and ecological values of traditional saltworks. In order to ensure such compatibility, rehabilitation or restoration of these sites requires careful planning and a continuous feedback of information from research and monitoring (Petanidou, 2000; Crisman *et al*., 2009). Despite these and other potential uses, the loss of hyperhaline biotopes, especially of solar saltworks, attained 74% in Spain, 63% in Italy and 55% in Portugal during the two decades preceding a review by Amat *et al*. (2007).

In recent decades a number of research papers have addressed the role of saltworks as alternative or complementary habitat for aquatic birds and their potential to compensate coastal habitat loss or supplement the remaining natural wetlands (e.g. Britton and Johnson, 1987; Velasquez, 1992; Masero, 2003; Sadoul *et al*., 1998). In the Iberian Peninsula, most studies have been conducted in Atlantic coastal wetlands (Masero *et al*., 2000; Múrias *et al*., 2002) establishing conservation and management proposals (Velasquez, 1992; Masero, 2003; Sánchez *et al*., 2006), while only a few have been developed in the Iberian Mediterranean coast, evaluating the consequences of their abandonment (Paracuellos *et al*., 2002). More studies focus on the landscape-scale consequences of abandonment of large industrial saltworks and their effects on flagship species (e.g. Greater Flamingo; Béchet, 2009), than on the design, management and monitoring of small-scale restoration projects, notwithstanding that artisanal saltworks are not only in greater risk of abandonment or reclamation but can also be more easily restored. Water surfaces managed primarily for conservation are of great value and should be increased in number and diversity of bio-physical conditions (Múrias *et al*., 2002; Tavares *et al*., 2009).

The potential use of abandoned saltworks and other artificial wetlands as “green filters” or treatment systems for nutrient-enriched waters is a particularly relevant aspect in the management of CL complexes. It brings together issues like wetland restoration, environmental services, and biodiversity. Waterbirds can benefit from these wetlands providing increased suitable habitat, and society can benefit from both the environmental services they perform (regulation of hydrological regimes, water purification, e.g. Maltby, 1986; Everard *et al*., 2012), and from the multiple values and functions (recreational, educational, scientific, bioindicator) provided by their birds (Amat and Green, 2009; Wenny *et al*., 2012).

Aquatic birds as bioindicators

Waterbirds are valuable top-level indicators of habitat conditions in CLs. Changes in the intensity of human influence (impact and/or management) cause dramatic changes in the composition, structure and conservation value of their waterbird assemblages, whatever the temporal or spatial scales at which they are examined. Waterbirds respond to changes imposed by a variety of stressors, constituting warning signals against undesirable changes (Amat and Green, 2009; Mallory *et al*., 2007), and can be used as bioindicators both at suborganismic and at population-community-ecosystem level (Kushlan, 1993).

Among the biota of CLs and other transitional coastal waters, waterbirds are probably the more intensively and regularly monitored group, with the indispensable help of citizen science (Robledano and Farinós, 2010).

Their potential use as bioindicators has been widely discussed (Peakall and Boyd, 1987; Adamus, 1996; Green and Figuerola, 2003; Gregory *et al*., 2003; Stolen *et al*., 2004; Rönkä *et al*., 2005; Amat and Green, 2009). Although recent research on bioindication in coastal waters has encouraged more intensive monitoring of waterbirds (Hubina, 2008; Rönkä *et al*., 2008), in the particular case of CLs, birds have received little attention compared to other taxa (Robledano and Farinós, 2010) with few studies addressing waterbirds as integral components of the food webs (Acuna *et al*., 1994; Žyydelis and Kontautas, 2008). In man-made or restored wetlands, aquatic birds are the preferred biological group on which to base the assessment of their ecological value and conservation status. The presence or abundance of waterbird species or guilds, and the changes in their relative abundance, have proven effective proxies of the biological integrity of wetlands and are considered good indicators of the ecological status of a site and the environmental changes that may occur in it (Bradford *et al*., 1998).

Objectives

We use the results of two studies carried out in a representative Mediterranean lagoon complex of SE Spain (Mar Menor), to show the different physical scales and time spans over which waterbird communities change, using several parameters and indexes as signals of environmental change with a direct influence on biodiversity conservation.

On the basis of these studies we aim to accomplish the following objectives:

- ✓ To assess the utility of waterbird community monitoring as a tool for tracking ecological changes in CLs and their associated wetlands, in response to human influences at different scales, both spatial (site, lagoon basin, watershed) and temporal (short and mid- to long term)

- ✓ To evaluate the performance of different community descriptors and indexes as response indicators of ecological changes in these wetland complexes

- ✓ To integrate the results of waterbird monitoring programmes made at different spatial and temporal scales within CL complexes, into a general evaluation system of ecological integrity

- ✓ To devise a habitat restoration scheme based on the experience in waterbird monitoring of representative wetland types within the Mar Menor CL system, which maximizes biodiversity enhancement and environmental functionality of the restored and created habitats

The case study sites represent the extremes of a size gradient: on one hand, a small restored pond or sub-lagoon (Coterillo Pond = CP hereafter) with a few hectares of surface area, and on the opposite, the full littoral area of the Mar Menor CL (tens of square kilometres). They share the characteristics of being shallow hypersaline waters with a strong dependence on external influences, whether natural (atmospheric, hydrographic) or artificial (hydrological impacts, eutrophication, management). Particularly, the lack of proper management at the watershed scale (in the case of the main natural basin) or at local level (in the small artificial site) can cause a long-term habitat deterioration.

These studies complement those previously made in the deepest part of the main lagoon basin, in combination with which a general framework for habitat restoration and management can be set.

Study area and methods

General characteristics of the study area

The Mar Menor Lagoon complex (Figure 1) is a 135 km^2 hypersaline coastal waterbody surrounded by a number of wetlands originated through several terrestrial and littoral processes leading to the isolation of small secondary basins, historically modified by man for different purposes (mainly salt production). Succession toward more terrestrial conditions due to sediment loading into these secondary basins, has led many of them to evolve into shallow wetlands or coastal steppes, where artificial saltpans have been created through dyking, channeling of freshwater flows and water pumping from the main lagoon.

As a result the Mar Menor complex is now a mosaic of wetland habitats with an ample variety of depths (from salt steppes without a water table to the deepest sections of the lagoon with several meters) and sizes (from small ponds of a few hectares to the main lagoon of more than a hundred square kilometres). According to these morphological gradients, the range of environmental impacts and conservation issues is also extensive and conforms to the biophysical characteristics of each wetland or habitat type. The main lagoon basin has remained relatively unchanged in its basic features, although dramatic changes have been reported in several parameters and habitat characteristics. Among its main threats are the external influences (land use changes in the watershed, hydrological alterations), and the modification of littoral habitats due to urbanization, coastal works, and other sources of disturbance (Álvarez-Rogel *et al.*, 2006; Velasco *et al.*, 2006; Carreño *et al.*, 2008; Robledano *et al.*, 2010; Álvarez-Rogel *et al.*, 2011). Overfishing and eutrophication from sewage and agricultural pollution threaten the biotic communities of the lagoon, as well as the lixiviation of metal-contaminated waste originated in past mining activities (Marín-Guirao *et al.*, 2005; María-Cervantes *et al.*, 2009).

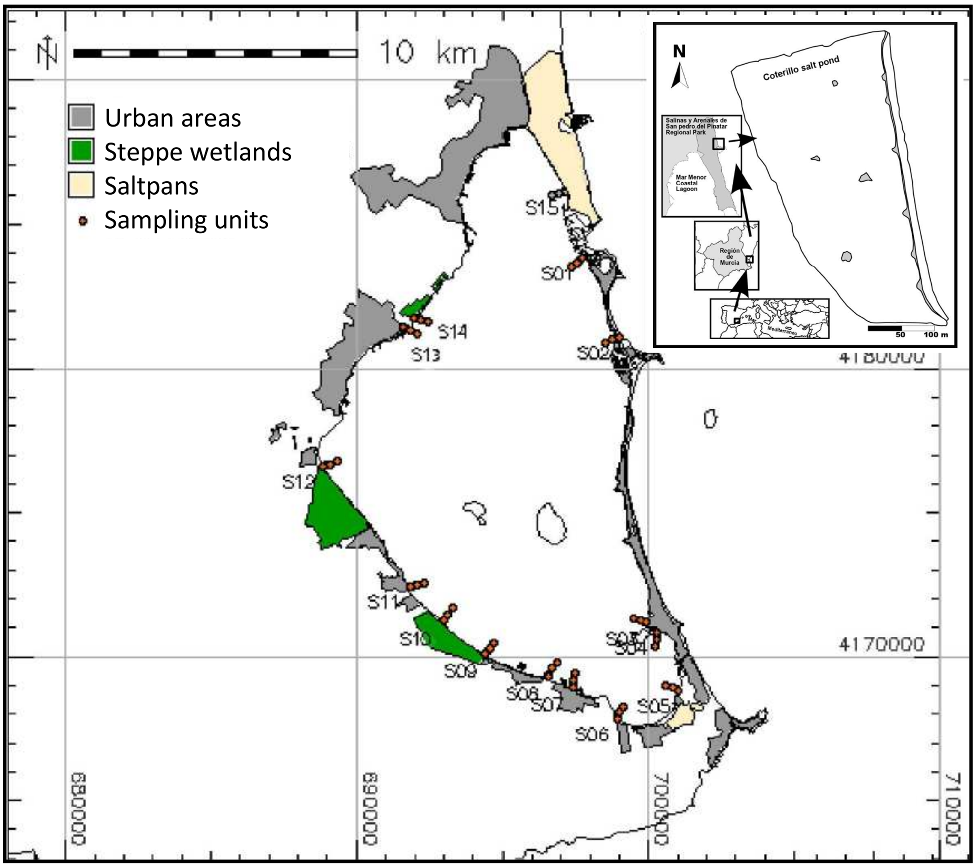


Figure 1. General map of the Mar Menor Lagoon and details of the study sites: Coterillo pond (upper right, including general geographic setting), and lagoon shoreline (main map) showing the 15 bird sampling units and the main associated wetlands: active saltpans (salinas) and saline steppe areas and other phreatophytic formations.

Associated wetlands around the lagoon have suffered from urban encroachment, tourism development and dumping of waste, sometimes invading physically their basins (particularly after the abandonment of salt production). Recreation pressure is usually higher in these areas due to easier access, although water-based activities also threaten even the remotest parts of the main lagoon. In recent decades most peripheral wetlands have been declared as protected areas, but this status does not always keep them safe from external influences like freshwater flows, contaminants and nutrients reaching them from the surrounding areas. Human pressure has also increased as the boundaries of these protected areas are gradually encroached by intensively developed areas.

Case study sites and sampling points

Figure 1 shows the location of the study sites around the Mar Menor. As previously said, two sites were selected, representing the extremes of a size gradient: on one hand, a small restored pond (Coterillo) with a few hectares of surface area, and on the opposite, the full littoral area of the Mar Menor CL (tens of square kilometres).

Within the evident size difference, these sites share the main characteristic of being shallow water areas, highly dependent on external influences, wether natural (atmospheric, hydrographic) or anthropogenic (hydrological impacts, eutrophication, management).

Coterillo pond (CP)

It is a small restored wetland created from an abandoned section of the salt production complex (salina) of San Pedro del Pinatar. The saltworks are included in the Regional Park “Salinas y Arenales de San Pedro del Pinatar”, a system of halophytic and psammophytic hábitats located in the North of the Mar Menor Lagoon (Figure 1).

The pond was excavated in the late 1950s with the objective of increasing the evaporating surface within the saltworks, abandoning its use in mid-1980. At the end of this decade, the pond was restored as part of a project for the conservation of *Aphanius iberus* (Valenciennes, 1846), funded by the EEC Regulation MEDSPA (Special Action Programme for the Mediterranean, 1986-1991).

Aphanius iberus is a Cyprinodontid fish endemic to the Iberian southeast, which usually does not exceed 6 cm in total length. Its diet is omnivorous, feeding mainly on crustaceans, insect larvae, detritus and algae (Oliva-Paterna *et al.*, 2006). The pond quickly became a valuable area for many waterbird species, including some appearing in Annex I of the Bird’s Directive (2009/147/EC, the codified version of Directive 79/409/EEC as amended). In the following years the management objectives were gradually reoriented towards an interpretative and educational function.

Among other functions performed by the pond, we should highlight the “green filter” role exerted over the past decades. The water supply of the pond has been a constant effluent coming from the experimental fishponds of a nearby research center (Marine Resources Center). This effluent carries both excretion products of fish raised in the fishponds and other organic debris, resulting in a general composition characterized by high concentrations of organic forms of P and N. Since its creation, the pond has undergone important changes in both morphological and physicochemical features, which could be related to the nature and composition of the effluent and the absence of any explicit management.

The study area consists of three basic units: the pond main basin (hereafter, the pond), a canal that runs longitudinally along its eastern shore, separated from it by a sand levee, and an adjacent salt marsh located to the north of the basin, created by subsurface diffusion of water from the main basin and therefore closely dependent on the pond in its water regime and water quality. The pond and canal are not fully independent since there are artificial connections between them, which add to the subsurface connection under the levee.

The pond surface is 8.2 hectares and it has an approximately trapezoidal shape (Figure 1). It contains four flat islands that are usually occupied for shelter and/or nesting by some bird species. The eastern canal is designed to help regulate the flow of water. Water enters the southern end of the canal and is discharged to the sea from the northern end through a pipe (which has not been permanently operational).

There are floodgates between the pond and the canal, but there has been no clear pattern of flow management. Moreover, the canal has suffered frequent breaks, maintaining a permanent connection with the main basin, although it has generally behaved as a semi-independent hydraulic compartment.

In the near future there are plans to reconnect the pond to the salt production network of the industrial saltworks, which implies the redirection of the effluent from the experimental fishponds to the sea, bypassing the pond system.

Mar Menor shoreline (MMS)

To conduct census on which to assess the indicator value of waterbirds, it is important to choose sampling sectors with similar baseline conditions (physical features), but with differences in the composition of their communities large enough to reflect the impacts they support. These impacts depend on seasonal changes in human pressure, as well as on the spatial distribution of the terrestrial factors affecting littoral waters and adjacent terrestrial areas (e.g. urban development or intensive agriculture, and the impacts induced by them). Sectors directly influenced by artificial connections of the lagoon with the open sea, will differ from those which are not hidrologically disturbed, as well as those in more natural landscape contexts will do so from sites bordered by artificial structures (harbours, seaways or urban fronts). Therefore, we chose a total of 15 points, distributed along the entire coastline of the Mar Menor to reflect a representative sample of the shore, which will be detailed below.

The low lying nature of the lagoon shores, with ample extensions of shallow nearshore waters (gently sloping to the deepest sections, rarely under -6 m), allow the comparison of sampling units on the basis of land-use and impact features superimposed on the general configuration of this sedimentary coastal area.

Each unit (centered on one of these points) needs a configuration that guarantees coverage of a representative and equivalent area, both on nearshore and deeper water sections. The shore length of each unit extends to approximately 500 meters, from which we established four parallel bands whose external boundary is at 100, 250, 500 and 1000 meters from the shore. The outer edges of the bands were gradually more regular polygons to close as approximately rectangular sections, covering a total area around 50 ha. This allowed us to establish 60 census cells (4 bands x 15 points). The minor differences in size and shape between units and cells reflect differences in the morphology of the shore and the need to establish ground references to facilitate recognition of their limits.

Among the 15 points (=units), there are natural and urban or semi-urban sectors. Some include or lie close to a recreational or fishing harbor; others are located next to the mouth of one or several ephemeral watercourses (locally *ramblas*) or drainage canals. Moreover, some sampling units were fringed by halophytic vegetation, reedbeds or other natural communities, others were bordered by agricultural land or urbanizations, and some had combinations of both. Regarding water quality, clear water sections contrasted with areas affected by eutrophication to a different extent. Sampling points located east of the lagoon, have marine (Mediterranean) influence unlike those located on the western, continental side, whose hydrological and hydrochemical influences are land-based activities (e.g. fertilizer application).

Biological data gathering

Waterbird communities were surveyed in both study sites through monitoring programs covering different time spans and with different periodicity. In Coterillo Pond, bird census started in October 1999 and have been extended until May 2010, being carried out mostly by G. A. Ballesteros, head of the “Programa de Seguimiento Biológico en Humedales de la Región de Murcia” (Murcia Region’s Wetlands Biological Monitoring Program), funded by the Autonomous Government. During this same period census of breeding pairs were conducted in the pond, including also the adjacent saltmarsh in the year 2000. The results of such census have been analyzed separately and in conjunction with those made in the pond (thereby constituting the pond+saltmarsh complex).

The methodology used in the census was the total direct count through comprehensive visual surveys in each of the water bodies, with suitable optical equipment (binoculars and spotting scope), from the banks and vantage points (Tellería, 1986), preferably in the early morning, trying to concentrate census effort in time, to improve the homogeneity of the data. The methodology used in the censuses traditionally applies equally to all groups included under the broad definition of waterfowl set by the Ramsar Convention (birds ecologically dependent on wetlands). For the study area, this refers essentially to open water bodies without dense associated emergent vegetation, making it possible to determine precisely the population of waterbirds.

The estimates of the number of pairs of waterbirds have been made by combining the direct counting of the breeders and young of the year, with the search and count of nests, eggs and chicks. For such estimates to be as accurate as possible, the criteria used as evidence of breeding have been those established by the European Bird Census Committee (EBCC, see e.g. Hagemeijer and Blair, 1997), adapted by Martí and Del Moral (2003). Besides birds, another biological feature was assessed in the pond: the general population and conservation status of *Aphanius iberus*. The aim was to determine relevant changes both in the density and in the distribution of this fish species within the pond and its peripheral canal, as a response indicator of the ecological changes that the system was undergoing. Through the characterization of population status we also tried to assess the pond’s performance and adequacy as a habitat for the reintroduction of the species. We considered this endemic fish as an indicator with different ecological requirements than waterbirds, although related to them since it represents a potential food resource, especially for wading birds and terns.

Fish sampling was made through the placement of 13 Minnow traps over a 24 hour period, the traps being distributed uniformly along the canal and the west and east margins of the pond. This type of trap, devised to capture small fish, is used commonly for estimating population size on the basis of capture/recapture ratios, but in our case we only counted adults and juveniles *in situ*. It is a passive method in which the individuals captured are not removed (Bryant, 2000).

In the Mar Menor Shoreline, bird censuses were performed during three winters (October 2006 to March 2007, October 2008 to March 2008 and October 2009 to March 2010) and during two summers (April to September 2007 and April to September 2008). The surveys were carried out by P. Farinós.

During these periods, direct counts of individuals of the wintering and summer species selected (all waterbirds except shorebirds, since the target assemblage was that using the littoral lagoon’s waters) were accomplished in each of the sampling points. The counting was done with a spotting scope mounted on a tripod and binoculars. Fixed visual references were used on land and in water to estimate distances on which to determine the limits of the whole unit and its bands.

Recording of environmental variables

Coterillo Pond was surveyed for some morphological and limnological characteristics, although only three times since its creation (1995, 2007 and 2010, the two latter within the period covered by bird census). Additional qualitative observations were combined with the results of these surveys to make a general description of the main macroscopic phases in the evolution of the pond.

Both the pond and canal were morphologically characterized through the measurement of their perimeter and depth. This was made with a GPS (Trimble GeoExplorer XT) for recording the perimeter and by setting seven transects in the pond and an equal number of points in the canal for measuring depth.

Along each transect three points (A, B and C) were marked to record depths, one in the center of the basin and the other two at 2,5 m of the east and west banks, respectively. Due to the shallowness of the pond and the prevailing low depth values, measurements were made manually with a rigid steel tape.

The physicochemical characterization of the pond and canal was made with a Multi 340L/SET probe, the sampling points corresponding to those used to measure depth but adding a new point in the pond, located in the vicinity of the place where the effluent was discharged. In each point we recorded the temperature (ºC), pH, conductivity (µS/cm) and salinity (g/l) of the water. To determine the concentration of the major nutrients (nitrate, nitrite, ammonium and phosphate) in the canal and pond, 15 samples were taken in high-density polyethylene bottles. The results of the laboratory analyses were expressed in parts per million (ppm) for nitrite, ammonium and phosphate, and in parts per billion (ppb) for nitrate. Nutrient concentrations give information on the productivity of the system and on the major biochemical processes occurring in it.

In the lagoon shoreline, the 15 units (centered on the census points) were characterized on the basis of the general landscape setting of the area (natural or artificial). A general description is given in Table 1. Bands within units represented a gradient of increasing depth extending offshore.

Unit	Description
1. La Encañizada vieja	Located in the North part of La Manga (sandbar closing the lagoon), adjacent to an urbanization; a stretch of natural sandy coast with halophytic scrub and shallow flooded areas belonging to the *golas* (natural canals through the sandbar) managed traditionally for fishing with an ancient catching system (*encañizadas*) restored in this area but abandoned or destroyed elsewhere in the CL
2. Playa Estacio	Area located near the Tomás Maestre harbour in the northern part of La Manga. Sandy and gravel gently solping beach of predominantly urban nature, with blocks of flats adjacent to one of its ends. There is a remarkable presence of fishing nets in the adjacent part of the lagoon.
3. Isla del Ciervo (North)	Stretch of semi-urban steep rocky shore (western side of the volcanic hill Monte Blanco) and its adjacent lagoon area. It is located in the central part of La Manga, facing the north side of Isla del Ciervo and being limited by other nearby islands (Redonda and Sujeto).
4. Isla del Ciervo (South)	Adjacent to the previous unit and located in the old artificial isthmus (now removed) of the Isla del Ciervo, oriented towards its southern side. Gently sloping sandy beach close to an urban area and next to a harbour.
5. Canal de Marchamalo	Semi-urban sector located in the southernmost end of La Manga, adjacent to the Marchamalo coastal saltworks. It is a low sandy beach surrounded by halophytic scrub impacted by landfills and adjacent to residential buildings.
6. Playa Honda	Semi-natural sector next to a coastal urbanization. A stretch of sandy coast, bounded by an urban façade, although its general nature is that of a mixed urban-agricultural landscape. Intensive human use of the littoral waters in summer (mainly recreational activities).
7. Islas Menores.	Section of urban coast located between two harbours. Gently sloping sandy coast adjacent to a promenade bordered by a strip of low buildings on the terrestrial side. Affected by a continuous (baseline) stress due to the transit of boats from and to both harbours.
8. Punta Lengua de la Vaca (Desembocadura R^{a} Carrasquilla)	Natural stretch of stony coast (large sized material deposited by streams flowing into the lagoon) surrounded by agricultural landscape. The census point was located at the outlet of the Carrasquilla ephemeral watercourse.
9. Los Nietos	Sandy beach of fine particle size. The observation point was situated in the contact of the natural area (characterized with dense formations of helophytes), with the promenade belonging to a nearby urban sector.
10. Saladar de Lo Poyo	Natural sector within the wetland complex of Lo Poyo, a stretch of gently sloping sandy beach adjacent to large patches of psammophytic vegetation

	growing on littoral dunes, and of halophytic scrub surrounding ancient saltworks. The whole area is influenced by two ephemeral watercourses flowing into its northern and southern ends, respectively (responsible for the disposal into the wetland of large amounts of mining waste with high concentrations of heavy metals).
11. Los Urrutias	Integrated into a stretch of semi-natural sandy beach, bounded by a harbour and a residential area, but keeping a dense patch of reedbeds of medium height. Noteworthy is the presence of buoys of maritime demarcation near the shore, usable for certain species as resting places.
12. Desembocadura Rambla Albujón.	Stretch of natural, gently sloping sandy beach with patches of reedbed and salt marsh. Its littoral waters show eutrophic conditions due to the inflow of nutrients through the channelized outlet of the Albujón ephemeral watercourse.
13. Los Narejos	Located within an urban stretch of coast with a promenade and a dense urban façade behind. Noteworthy is the presence of a pier on shore.
14. Playa de La Hita.	Stretch of natural, gently sloping beach into a dense belt of reedbeds; noteworthy is the proximity to disturbing elements of anthropic origin such as San Javier airport.
15. Salinas de San Pedro del Pinatar.	Sector of natural conditions within the Regional Park of Salinas de San Pedro del Pinatar. It is a stretch of gently sloping beach adjacent to several large saltpans of the industrial saltworks complex, which can be used by many species of waterbirds as a resting and refuge area.

Table 1. Description of the 15 bird sampling units of the Mar Menor CL shoreline

Community descriptors and statistical analyses

Coterillo pond

In Coterillo Pond, bird censuses were analyzed separately for winter (October-March) and Summer (April-September), two broadly defined phenological periods including, respectively, the true wintering and breeding periods, plus some transitional months in each case. As defined, these periods extended from the autumn passage and the building of the wintering community to its gradual departure, and from the spring passage and arrival of breeders to the post-nuptial period of dispersion of adults and young reared in the study area. “Winter” gathers together mainly birds having bred in more northern latitudes in passage or staying in the Mar Menor CL system in the cold months, while “Summer” does so with migrating birds having spent the winter in more southern areas plus those that settle in the study area as breeders.

With the aim of detecting changes in the conservation value of the bird community, we calculated two multispecific indexes adapted from Pons *et al*. (2003), and previously used in other wetland habitats (Robledano *et al*., 2010). In the SPEC index, species are classified according to the ‘Species of European Concern’ categories established by BirdLife International (2004). In the LRAE index, bird species were ranked regarding their threat status (extinction risk) as reflected by the Spanish Birds Red Book (Madroño *et al.*, 2004), in which the species are evaluated against the IUCN Red List categories at the national level. Each category within these ranks was given a numerical value that was used to compute the index (Table 2).

Numerical value	**SPEC**	**LRAE (Spanish Birds Red Book)**	
4	SPEC 2	VU	Vulnerable
2	SPEC 3	NT, LC, DD	Near Threatened, Least Concern, Data Deficient
0	NON-SPEC	NE	Not evaluated

Table 2. Numerical values attributed to bird species ranks within the SPEC and LRAE categories, used to compute the respective indexes of conservation value

The values are attributed in a geometrical progression of increasing threat status or conservation concern (Pons *et al*., 2003). The value given to each species is multiplied by its abundance (logarithmically transformed), and the resulting values are summed across the community to generate a single index score for each year studied (Pons *et al*., 2003; Paquet *et al*., 2006). For breeding birds, total species richness and total number of breeding pairs were computed for each year. The total number of breeding pairs was split into three spatial divisions (pond, adjacent saltmarsh and the whole Regional Park to which the former belong), in order to explore the potential dependencies among them when interpreting numerical changes.

This was done through Pearson correlation analysis between Regional Park and pond, pond and saltmarsh, and Regional Park with the ensemble pond+saltmarsh. Finally, the waterbird community was analyzed on the basis of the distribution of its abundance among four functional groups or guilds, a concept widely used in community studies to refer to assemblages of species with ecological similarity not necessarily translated into taxonomic affinity (Wiens, 1989). Each guild can thus incorporate species from more than one family (and even from higher distinct taxonomic divisions).

✓ Ducks (*Anatidae*): Represented in CP only by one family and virtually by a single species, *Anas acuta*, whose trophic amplitude (feeds mainly on plants but also insects, worms and occasionally mollusks; Gordon *et al*., 1998), allows it to be considered as a generalist, i.e., capable to live in a variety of habitats and not specialized in a particular food source (Olmos and Boulhosa, 2000). In the context of CP succession, these species are globally regarded indicative of mature or stressed stages.

✓ Wading birds (also referred to as ‘waders and allies’): Birds generally large in size that feed in shallow waters, searching for food in the sediments or filtering water with specialized bills, or catching fish and other aquatic fauna. It includes three families (*Threskiornithidae, Phoenicopteridae* and the richest one, *Ardeidae*). Grey Heron *Ardea cinerea* (family *Ardeidae*) and Flamingo *Phoenicopterus roseus* (*Phoenicopteridae*) are considered specialists, i.e. species adapted to relatively narrow ecological conditions (Olmos and Boulhosa, 2000). In the context of the CP system, they are regarded as indicative of relatively undisturbed conditions.

✓ Shorebirds: Birds that feed in tidal areas and other hábitats experiencing fluctuations that expose or make available productive sediments and shallow waters. They explore marshes, beaches, rocky coasts and wetlands (freshwater or salty), searching for invertebrates (insects, worms, small crustaceans). In CP several families (*Recurvirostridae, Burhinidae, Haematopodidae* and mostly *Charadriidae* and *Scolopacidae*) are represented. This guild is particularly sensitive to wetland degradation, with dramatic variations in their abundance and richness occurring along with changes in the ecological conditions of their habitats. This causes them to be generally considered as specialists (Hayman *et al*., 1986). In the particular case of CP, these species are also considered indicative of undisturbed stages.

✓ Gulls and terns (families *Laridae* and *Sternidae*, also referred to as ‘gulls and allies’): are omnivore species that often include in their diets, besides fish, refuse and carrion. Thus, they are less sensitive tan other guilds to degradation, pollution or human presence in their habitats, and can be considered opportunists or generalists depending on the species (Buzun and Mierauskas, 1987). In CP, they are globally regarded as indicators of mature or stressed stages (i.e. eutrophic, polluted conditions).

The change of waterbird abundance, Shannon Diversity, SPEC and LRAE indexes, were analyzed with the R Statistical Package (R Development Core Team 2007), using the non-parametric Kruskal-Wallis test and establishing the statistical significance at $p = 0.05$ in all tests. The Shannon index depends on the number of species (richness) and also on the uniformity of their contribution to total abundance (number of individuals of each species).

The main temporal phases within the survey period (between which relevant changes in the composition of the waterbird community occurred) were established through multivariate ordination techniques. These methods can be used to describe the degree of similarity (and thus to differentiate) among temporal or spatial subdivisions, in terms of the identity and relative abundance of the species (Lopes *et al.*, 2005; Evangelopoulos *et al.*, 2008; Zárate-Ovando *et al.*, 2008; Sánchez *et al.*, 2006). Data were normalized (log(x +1) transformation) and submitted to Multidimensional Scaling (MDS; Clarke, 1993) implemented through the Primer 6 software (Clarke and Gorley, 2006), to group years on the basis of their similarity and hence identify the main phases of change and their indicator species. This should allow interpreting phases as a reflection of stages in the succession of the pond ecosystem, with characteristic waterbird assemblages.

A subsequent Indicator Value Analysis (IndVal) based on the Monte Carlo Test, performed with the PC-Ord Software (McCune and Mefford, 1999), allowed to identify representative (indicator) species for any year or group of them (Dufrene and Legendre, 1997), i.e. for the temporal phases recognized by the MDS.

Mar Menor shoreline

In the Mar Menor shoreline, like in the case of CP, two data sets were analyzed separately: winter and summer censuses. Three community variables, the mean annual abundance, the mean monthly abundance (between years) and the Shannon-Wiener index were calculated. In order to detect changes in the conservation value of the waterbird community, SPEC and LRAE indexes were also calculated.

The spatial and temporal variability of structural and conservation indexes was analyzed, using “season” and “year” as temporal factors to test for differences between winter and summer community and between years (within seasons), respectively. As in CP, the nonparametric Kruskal-Wallis test was used to determine the significance of these differences. “Basin” and “band” were also used as factors, to detect spatial variations in community structure and organization. “Basin” was used since in the Mar Menor lagoon two main sectors or *basins* with different hydrological characteristics are recognized (Pérez- Ruzafa and Marcos, 2003), which may determine also different particular conditions for waterbirds. Factor “band” was considered because each species may exploit different areas of the depth gradient depending on their morphological characteristics. Again, statistical significance was set at $p = 0.05$ for all analyses.

Multivariate ordination methods were also applied to shoreline waterbird data, to describe the degree of similarity (or difference) between the temporal or spatial subdivisions (on the basis of previously described factors), in terms of relative abundance and species composition.

After data normalization through a log (x+1) transformation, a Multidimensional Scaling analysis (MDS) was performed (Clarke, 1993) with the software Primer 6 (Clarke and Gorley, 2006). Subsequently an Indicator Value analysis (IndVal) based on the Monte Carlo test was implemented through the PC-Ord software (McCune and Mefford, 1999). This analysis allowed to recognize indicator species with statistical significance for each class within each grouping factor (basin, band and classes inferred from calculated rank values of LRAE and SPEC) that is integrated into the analysis (Dufrene and Legendre, 1997). Qualitative classes of conservation value were inferred from rank values of conservation indexes and assigned to sampling units; then, units were grouped according to these classes and IndVal tests conducted to detect indicator species of each conservation class (separately for each season).

Results and discussion

Coterillo pond

The results regarding Coterillo Pond are presented in a sequence that starts with the discussion of its main morphological and limnological changes, followed by that of its population of *Aphanius iberus*, all referring to non-ornithological aspects and based on a short series of surveys. Then there is a general analysis of the change of the waterbird community (both in the winter and summer period), and a specific one of the nesting populations, all sampled along a much longer and continuous time series. It concludes with an ordination-based approach to the temporal organization of the waterbird community, complemented with the recognition of indicator species that helps in their ecological interpretation.

Morphological and limnological changes

The results of the three surveys available (1995, 2007 and 2010) are presented separately for the pond main basin and the canal, since despite their connections they are considered semi-independent hydrological units (Table 3). For the canal there are no data for 1995.

POND (MAIN BASIN)	1995	2000-2006	2007	2008-2009	2010
TEMPERATURE (ºC)	25.36	No data	23.25	No data	25.90
pH	7.95		4.88		6.75
AVERAGE DEPTH (cm)	27.93		23.67		35.89
AVERAGE SALINITY (g/l)	22.87		58.42		49.33
NITRITE (mg/l)	0.002		0.0125		0.00001
NITRATE (mg/l)	2.45		<0.2		0.00003
AMMONIUM (mg/l)	0.04		<0.06		0.00108
PHOSPHATE (mg/l)	0.08		<0.2		0.00018
ADJACENT CANAL	1995-2006		2007	2008-2009	2010
TEMPERATURE (ºC)	No data		23.53	No data	25.57
pH			5.54		6.76
AVERAGE DEPTH (cm)			24.83		37.33
AVERAGE SALINITY (g/l)			43.70		38.85
NITRITE (mg/l)			0.010		0.0120
NITRATE (mg/l)			<0.2		N.D.
AMMONIUM (mg/l)			<0.06		3.36275
PHOSPHATE (mg/l)			<0.2		<0.002

Table 3. Morphological and limnological changes detected in the surveys made in the pond (3 periods, upper section) and in the canal (2 periods, lower section)

Taking as a reference situation the state of CP in 1995, in 2007 the features indicative of a degraded or stressed state had worsened: lower depth, increase in salinity and excess primary production, as well as a decrease in pH. All this seems related to the evolution of the system towards an apparent eutrophic state following nutrient loading in the water supply, and in the absence of any explicit management of factors affecting the system productivity (water residence time, internal circulation, depth management, removal of decomposing algae, etc.). The pond had evolved into a state of lower hydrological and ecological quality, far from the original objectives for which this artificial wetland was created. This trend was reversed in 2008 with the opening of the communication of CP to the sea, causing a *rejuvenation* of the system and a gradual return of the different physicochemical parameters of the water towards less stressed, baseline conditions. This new state (deeper water, more neutral pH and low salinity) could not be confirmed until the 2010 survey.

With regard to nutrients, the waste from the Marine Resources Center which is discharged with the effluent into the canal, is composed primarily of fish excreta, uneaten food and remains of other products used in fish culture. This waste, rapidly increasing nutrient concentrations (total phosphorus, ammonia, organic nitrogen) and organic matter content in the sediment and water column, decreases the concentration of dissolved oxygen and raises the general trophic status of the water body (Tartarotti *et al*., 2004; González-Félix and Pérez-Velázquez, 2006; Urakawa *et al*., 2006), wich results in the eutrophication of the system. In wetlands, the process of decomposition (mineralization) occurs at a significantly lower rate in the sediment, due to the prevalence of anaerobic conditions in most of its profile.

Consequently, a considerable accumulation of organic matter can cause mismatches in the system (Clymo, 1983). The continuous discharge of nutrients, mainly phosphate and nitrogen, stimulates the growth of algae and other photosynthetic aquatic life, accelerating eutrophication of the receiving environment, significantly reducing the concentration of dissolved oxygen and producing undesirable changes in aquatic populations (Knobelsdorf, 2005; Ruiz, 2008).

Aphanius iberus

Regarding the population of *Aphanius iberus*, it should be noted that in this type of wetlands the fish community displays a very low species diversity, a fact that can be decisive for the stable presence of this endemic species. Usually only three species of fish (*Anguilla anguilla*, *Mugil cephalus* and *Atherina boyeri*) coexist with this small cyprinodontid. In 1995 and 2007 only *Mugilidae* were recorded in CP. Thus, it is likely that the populations of *Aphanius* of CP benefit locally from reduced interspecific competition (Oliva-Paterna and Torralva, 2008). However, other abiotic factors may be determining the settlement of *Aphanius* populations, for example, the absence of hydrodynamism in these closed water systems (Smith and Able, 1994).

In the absence of a continuous monitoring of the population of *Aphanius iberus*, on the basis of the numerical estimates available (revealing a high density and an important number of pregnant females), the species can be attributed to an apparently good conservation status, as already observed by Farinós and Robledano (2007).

Two conclusions can be drawn from the population surveys made in the two periods studied (Table 4). First, at a general level, the proportion of females relative to males is quite high. In other studies (Fernández-Delgado *et al*., 1988; Granado-Lorencio, 1996), some populations show a numerical dominance of females over males. This dominance may result from a differential response of sexes to reproductive stress and sexual maturation. The significant dominance of females in these fluctuating environments, could be interpreted as the basis of a blooming response allowing effective population recovery. This species presents a life strategy that enables it to live in environments with significant intra and inter-annual fluctuations in environmental parameters.

	POND		**CANAL**	
	2007	**2010**	**2007**	**2010**
MALES	58	157	176	40
FEMALES	249	691	492	158
YOUNG	1	50	17	7
TOTAL CATCH	308	898	685	205

Table 4. Population indexes (captures) of *Aphanius iberus* in 2007 and 2010.

The general pattern of this strategy would be characterized by a low number of age classes, high growth rates and a short reproductive life, but with an extended breeding season (these aspects have not been confirmed but fit with the fluctuations known to occur in this ecosystem).

This is a typical strategy of an opportunistic species (Winemiller and Rose, 1992) adapted to extreme and unstable environments. Second, the distribution of *Aphanius iberus* in the system changed from 2007, when most of its population was concentrated in the canal, to be located preferently in the basin in 2010. This change may be due primarily to improved physical and chemical parameters, especially pH and probably dissolved oxygen.

Waterbird community

The long term variation of the composition, structure and conservation value of the waterbird community (Figures 2 and 3) is described by the changes in the relative abundance of bird guilds, in the species richness and diversity (total and intra-guild), and in the scores of the ornithological indexes (SPEC and LRAE).

The contribution of shorebirds to the total abundance remained relatively constant until the winter 05/06, when it markedly decreased possibly coinciding with the time when limnological conditions in CP started to deteriorate. As for ducks, they showed a reverse trend to that of shorebirds, since until the winter 05/06 they kept contribution to total abundance certainly modest (except in the winter of 2003). From 2005 it increased significantly, anticipating an upward trend for the coming years. The guild of waders, suffered a similar process to that of shorebirds, since they initially showed a significant contribution to the total abundance that was subsequently reduced.

The Shannon-Wiener index (which combines richness and abundance) showed an increase in all guilds, from the minimum in the winter 07/08 (which follows the summer of 2007 when the pond had a moderately advanced state of eutrophication), but it cannot be said that there was a return to baseline diversity values in all groups. Shorebirds would be the guild that showed greater sensitivity to the changes experienced by CP. This was confirmed by observing the species richness, which for shorebirds fell sharply, while in the rest of guilds remained without major variations.

The general decline in the contribution of shorebirds to the abundance and in species richness, and to a lesser extent in waders and allies, resulted in a loss of specialists, in favour of an increase in opportunistic or generalist species (ducks, and especially gulls and terns), a signal of the loss or degradation in the ecological condition and habitat quality.

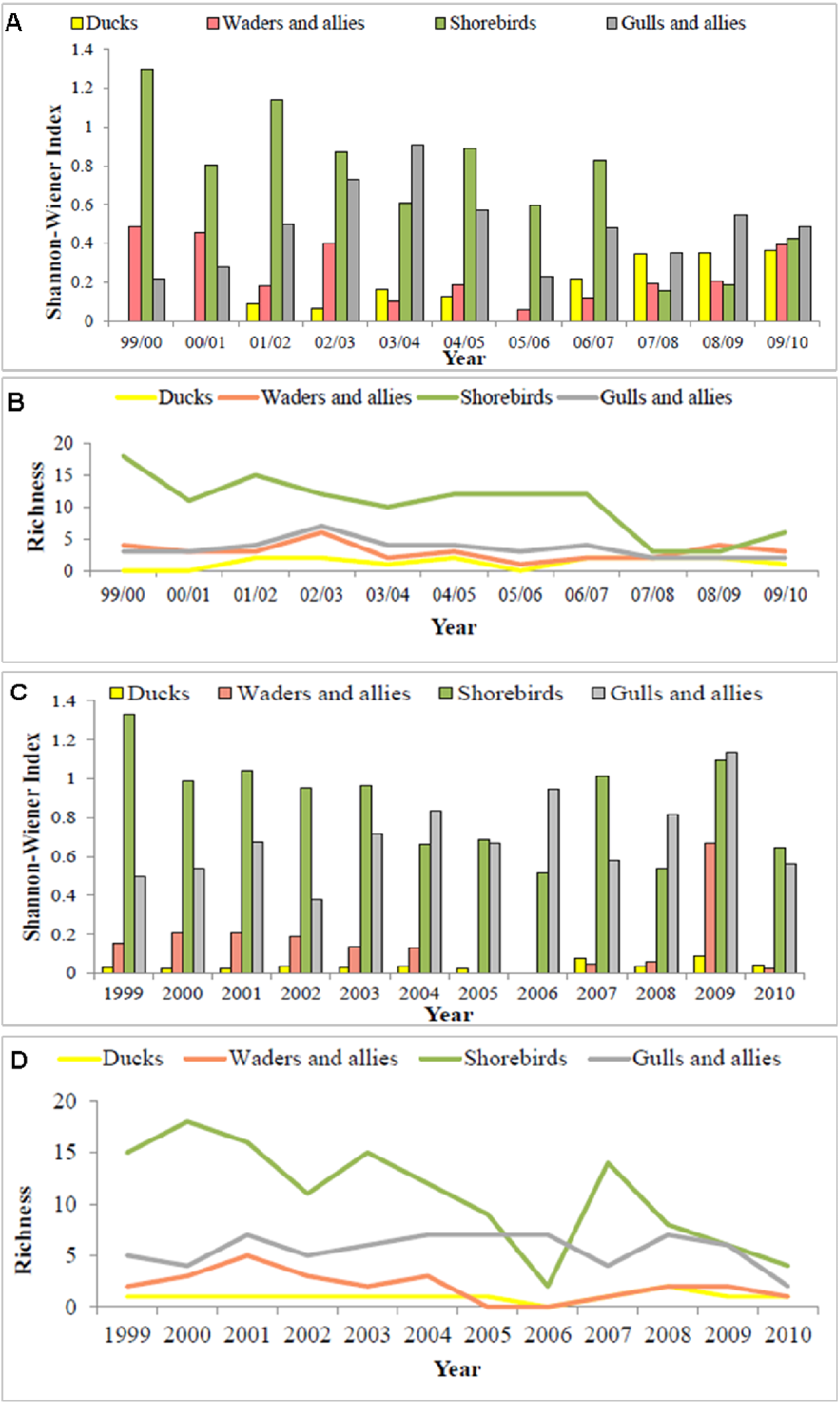


Figure 2. (A-D) From top to bottom : (A) Shannon-Wiener Index and (B) species richness within each waterbird guild in Coterillo Pond in winter; (C) Shannon-Wiener Index and (D) species richness of the same guilds in summer.

The contribution of waders to summer abundance was considerably reduced compared with winter, but remained constant throughout the entire surveyed period.

The two markedly dominant guilds (shorebirds and gulls and allies) showed a reverse trend. Based on the characteristics and preferences described for each guild, one might expect that the contribution to the abundance of waders in 2007 was very low, which did not occur. Two reasons can explain this fact. First, in 2007 the average depth was significantly lower than in the physico-chemical surveys of 1995 and 2010, resulting in a lower water table and therefore a larger surface of banks available for waders. Second, it could have been a certain delay in the response of birds to pond conditions, evidencing its effects in the winter 07/08 (and in the following summer of 2008) and not in the summer 2007 itself when the disturbance occurred.

The richness of shorebirds underwent drastic fluctuations. No data were available on limnological parameters for the summer of 2006, when very low numbers of shorebirds were recorded and when ducks and waders disappeared from CP. However, it is known that in May 2006, the communication with the sea was not open and the water level in the pond was very high, leaving little surface of exposed banks available for waders. No more information is known about the state of the pond, but the response of shorebirds is consistent with the situation of other guilds, especially wading birds. In that same year, the abundance of gulls and terns was well above the peak of waders, as did the guild’s species richness. The following year, the trend was completely reversed.

The frequent fluctuations in the waterbird community parameters, both in winter and summer, were in some cases difficult to explain due to the lack of a parallel monitoring of limnological and other environmental variables with similar continuity. In any case, the general trend observed in summer continued to be the gradual replacement of specialists by opportunists. Although gulls and terns kept a species richness around 4-7 along most of the study period, their contribution to total abundance increased every year, what could be a symptom of the system’s deterioration.

Figure 3 shows the variation of three of the parameters used to characterize the waterbird community: SPEC, total abundance and total species richness. The latter showed a decreasing trend along the census period with significant differences among winters (Kruskal-Wallis test). This trend is related to the disappearance from CP of many members of the shorebird guild, the most negatively affected of all those represented. Among the species lost are most *Scolopacidae* and some *Charadriidae*. In the last years surveyed, a member of the *Sternidae* (*Sterna sandvicensis*) is also lost.

As for total abundance, it showed higher values in the first half of the survey period, until the winter 04/05, when the peak of abundance was recorded. Subsequently, it started to decrease until it reached a minimum in the winter 07/08. The trend of the SPEC index was concordant with those of total abundance and species richness.

Again, two neatly defined periods could be distinguished: until the winter 04/05 there was a rising trend, and then a decrease until 07/08.

At the end of the study period, the index rose again due mainly to four species, each belonging to a different guild, being incorporated to the community: a duck (*Anas acuta*), a wading bird (*Phoenicopterus roseus*), a shorebird (*Tringa totanus*) and a gull (*Larus genei*). Although the situation seemed to have improved with respect to the previous assessment made in 2007, the community was still far from the composition it had in 04/05, as a result of the loss of shorebird species.

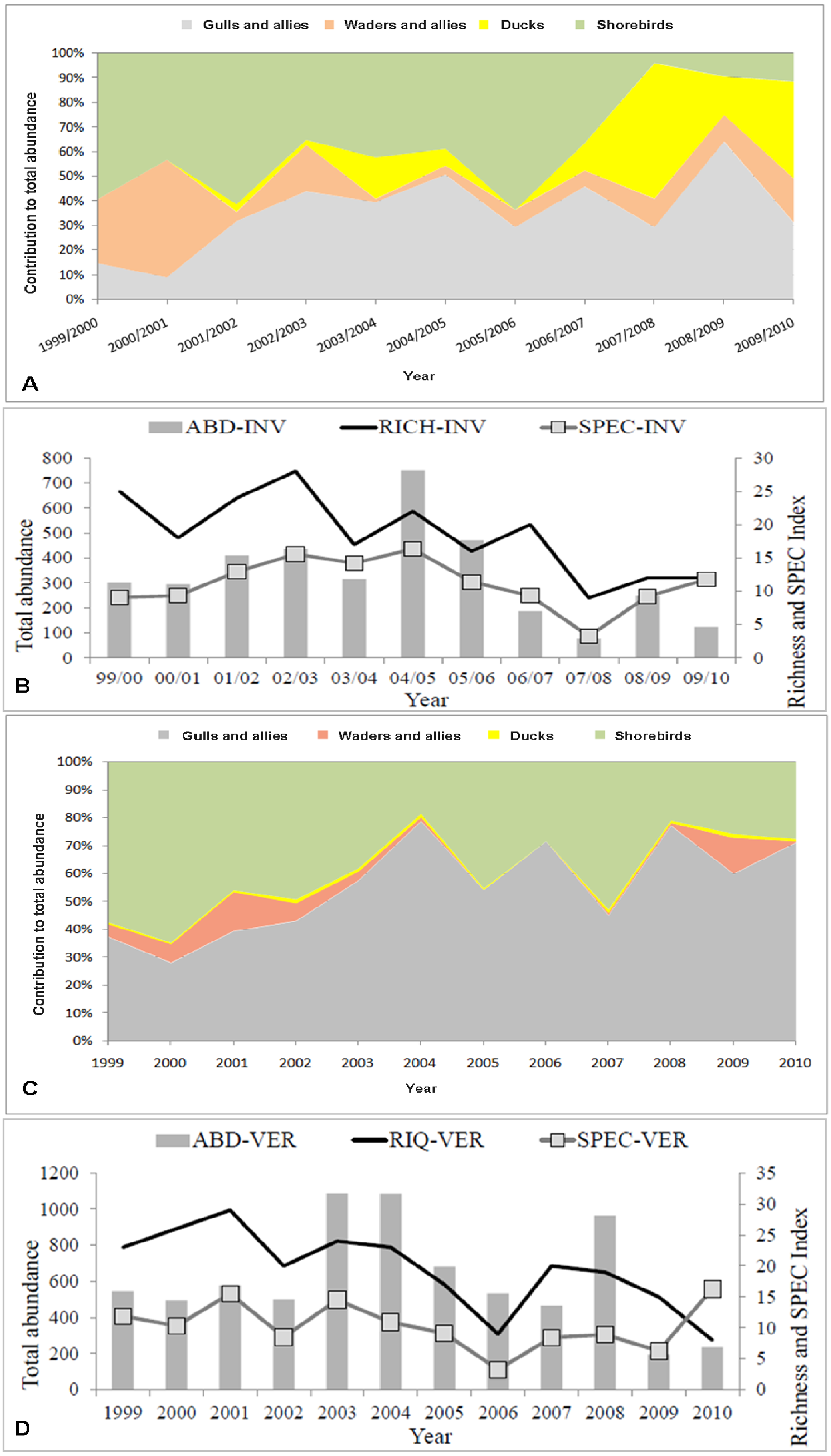


Figure 3. (A-D) From top to bottom: (A) Contribution to total abundance by each guild in winter; (B) Total abundance (left ordinate axis), Total Richness and SPEC Index (right ordinate axis) in winter (=INV); (C) Contribution to total abundance by each guild in summer; (D) Total abundance (left ordinate axis), Total Richness and SPEC Index (right ordinate axis) in summer (=VER)

Nesting populations

Among the nesting families, the contribution of shorebirds and gulls and terns can be highlighted. This responds to several factors concurring in the Regional Park (and to some extent also in CP).

First, both represent an appropriate breeding habitat (particularly due to the presence of bare sand levees and islets, beaches, accumulations of sea drift debris, etc.). Second, they exhibit a high availability of food for these guilds, like benthic invertebrates and fish. *Sterna hirundo* and *Sterna albiforms* specialize precisely in the capture of small fish (Cramp, 1988), and both the Park and CP have abundant populations of *Aphanius iberus*. Other *Sternidae* like the Gull-billed Tern (*Gelochelidon nilotica*) feed preferently on small reptiles and amphibians (Cramp, 1988), which are also abundant in the surroundings, especially the latter due to the development of a dense irrigation system in the nearby agricultural plain (Sebastián-González *et al.*, 2010). One of the most abundant species, *Larus ridibunus*, can also be favored by the proximity of the sea where it uses to benefit from fish discarded by fishing wessels (Goutner, 1994).

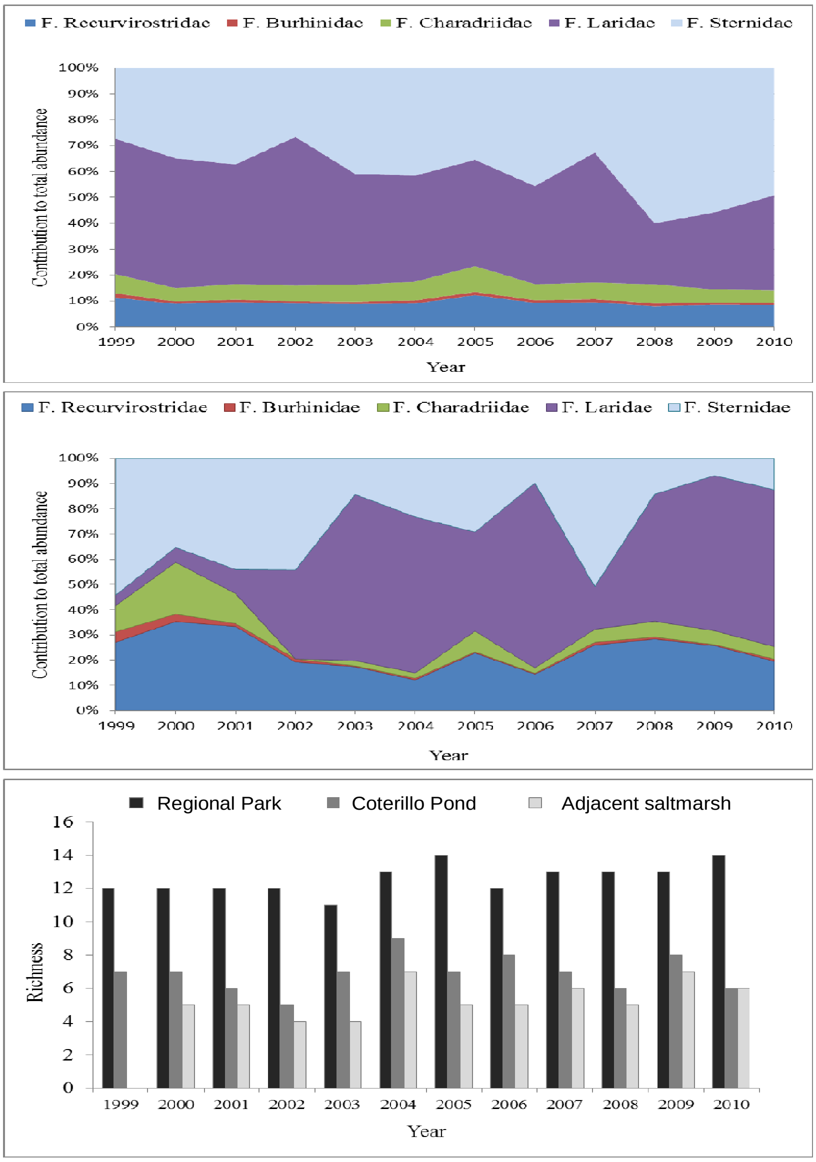
In regard to shorebirds, represented by *Recurvirostra avosetta*, *Himantopus himantopus* and *Charadius alexandrinus*, their diet is based mainly on crustaceans, insects and worms. Data from Velasco *et al.* (1995) support the presence of several invertebrate prey (*Diptera*, *Coleoptera*, *Amphipoda*, etc.) which make Coterillo Pond system (including the pond plus its adjacent saltmarsh) an attractive place for waterbird breeding in the context of the Regional Park to which it belongs.

Regarding percent contribution to total abundance, that of the families *Laridae* and *Sternidae* (gulls and terns) is the most remarkable, both in the Regional Park and in the Coterillo Pond+Adjacent Saltmarsh (CP+AS, hereafter) ensemble. The breeding community of the latter changed in a somewhat independent manner with respect to the general situation in the Regional Park, which depicts a response to the particular conditions prevailing in the pond and its tightly related adjacent saltmarsh (i.e., to the more fluctuant ecological regime of such environment, not dictated by the functioning of the industrial circuit of salt production as in most of the Regional Park).

In the ensemble CP+AS the terns seem to be replaced by more generalist species like *Larus melanocephalus o L. ridibundus* (virtually not represented in the early stages of the development of the pond ecosystem). This replacement is attributable to two factors: first, to the already mentioned deterioration of the initial ecological status of the CP, affecting negatively the specialist taxa. And second, the settlement of a very high number of pairs of Black-headed Gull *Larus ridibundus* (190 in 2003, 257 in 2004, 202 in 2006) in a relatively small area (8.2 hectares, but much smaller if the water table is not accounted for), could have prevented the settlement of other breeding species. The opposite trend is seen in the Regional Park, with an increasing contribution of *Sternidae* over *Laridae*.

The species richness of the breeding waterbird community of CP represents around 40-60% of that of the whole Regional Park. The species richness of the AS is similar to that of CP during most of the survey period. As it can be seen in figure 4, only *Recurvirostra avosetta* changes in relation to its variation in the whole Regional Park, while five species change with a certain degree of coupling between the two parts of the ensemble CP+AS.

From these observations, it was suggested that CP could probably function (in conjunction with other *satellite* wetlands, and obviously with the Regional Park) as a metapopulation-like system (Sebastián-González, 2011), at least for species behaving like *Recurvirostra avosetta.* At a smaller scale the fraction of birds present in AS would represent a subpopulation of those settled in CP, both behaving as a coupled system (for the remaining species in the table of figure 4). The trends observed in the censuses covering the whole Park, however, are not reflected in the numbers of breeding pairs of these species recorded in CP. This fact, and the lack of correlation between the Regional Park and the ensemble CP+AS for most species, indicates that the breeding community of the latter responds to a greater extent to local ecological conditions and does not mirror overall trends or ecological changes which are apparent in the whole of the protected area system.



	REGIONAL PARK WITH COTERILLO POND	REGIONAL PARK WITH COTERILLO POND AND ADJACENT SALTMARSH	COTERILLO POND AND ADJACENT SALTMARSH
Himantopus himantopus	-0.12	-0.10	**0.50**
Recurvirostra avosetta	**0.54**	**0.54**	**0.82**
Larus ridibundus	0.02	0.03	**0.73**
Larus michahellis	-0.22	-0.22	**1.00**
Sterna albifrons	-0.43	-0.34	**0.78**

Figure 4. Percent contribution to total abundance of the different families of waterbirds breeding in the Regional Park, and in the ensemble CoterilloPond+Adjacent Saltmarsh (upper graphs). Annual change in the species richness of the breeding waterbird assemblage in each of these subdivisions (lower graph), and correlation coefficients among subdivisions for the breeding populations of particular breeding species (table).

Temporal organization and indicator species

The ordination analysis on the basis of the census of the waterbird community, allowed recognizing three main temporal phases, clearly identifiable both in winter and summer (figure 5):

I) 1999-2002: The community was dominated by shorebirds except in one winter (gulls and terns, 02/03). SPEC and LRAE indexes showed high values although the peak values occurred later.

II) 2003-2007: Shorebirds end to be the dominant guild (in terms of their contribution to total abundance) in favor of gulls and ducks. Although the richness of the former guild continues to be the highest (except in the summer of 2006), it can be already said that shorebirds are receding and that the other two guilds alternate in the dominance of the community. Thus, it can be concluded that the specialist guild is being replaced by the more opportunistic or generalist ones.

III) 2008-2010: Waders no longer dominate in abundance and their richnes decreases too, but since the total richness of the community also decreases, the result is a co-dominance of shorebirds with wading birds, and larids, in 2008 and 2009, respectively

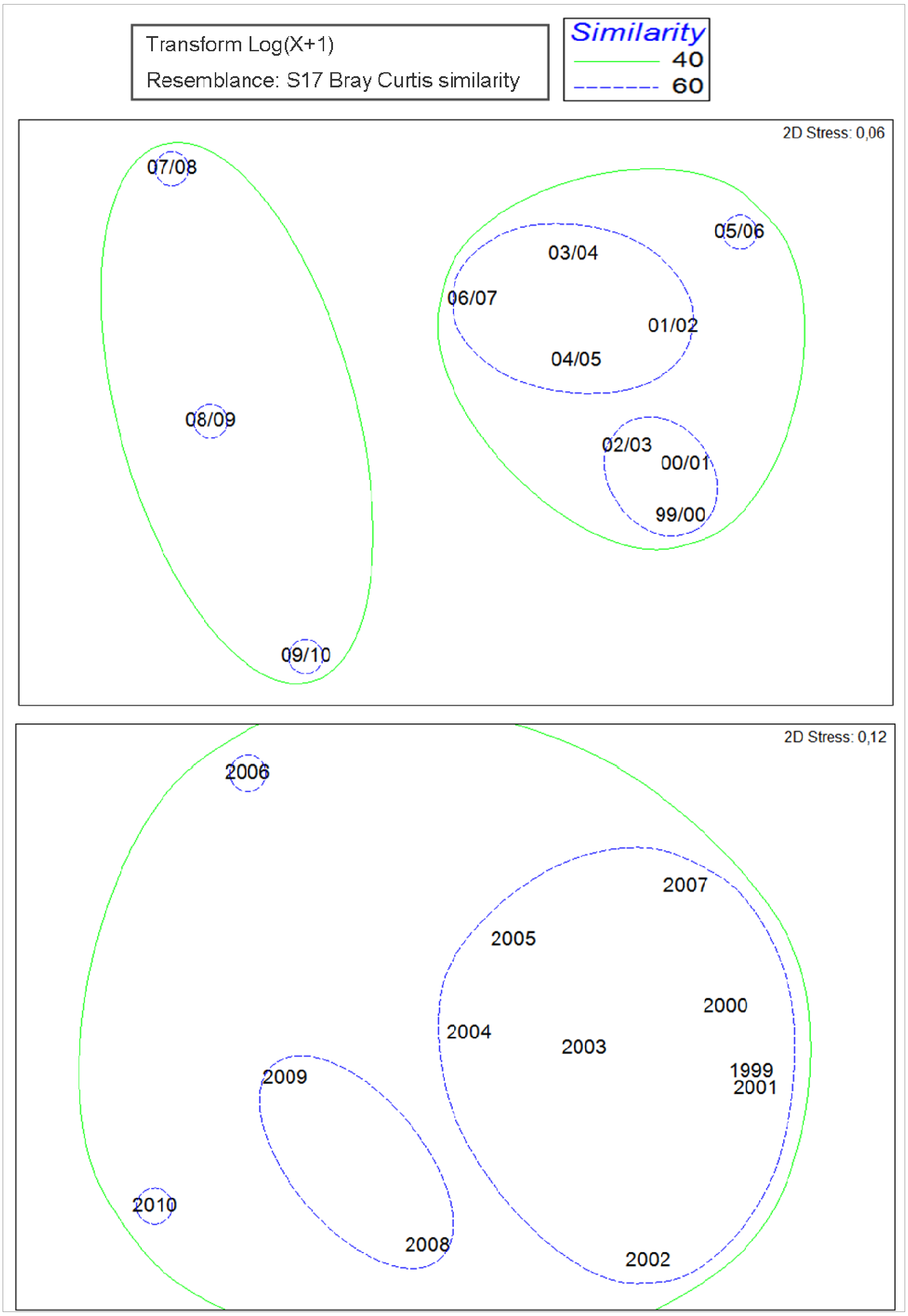


Figure 5. Multi Dimensional Ordination plot (up=winter; down=summer) identifying the main temporal phases on the basis of the census of the waterbird community

A summary of the results of indicator value analyses is shown in Table 5, which includes a qualitative assessment of the relative abundance and richness of species and guilds, as well as the definition of the three periods described according to the dominant strategy (degree of ecological specialization) among the waterbird community.

Regarding the winter periods, *Phoenicopterus roseus* (the flagship species of the Regional Park) appeared as an indicator species in that of 1990/00. Its relative abundance along the study period was quite variable, occurring in five winters (although more than half of its total numbers occurred in that of 90/00). Flamingos prefer brackish or saline, shallow water areas with depths usually less than 60 cm, although it can exploit depths of 90-105 cm (Johnson, 1983) to feed on carotenoid-rich prey (McGraw *et al.*, 2001). Such preferences suggest that in these winters and especially in 99/00 water depth was not very high in the pond, although limnological surveys are lacking and other ecological interpretation is not allowed.

It can be highlighted that both the species richness and the Shannon diversity index showed a decrease from 2004 to 2007, probably due to the eutrophication process described by Farinós and Robledano (2007).

During that period *Phoenicopterus roseus* was not recorded, suggesting lowered food resources and/or worsened feeding conditions for the species. Velasco *et al.* (1995) recorded the presence of crustaceans of the orders *Mysidacea, Isopoda, Ansipoda* and *Amphipoda*, potential prey of flamingos which tend to be sensitive to changes in pH and nutrient contents of the water. The eutrophication of CP could have reduced the density of these invertebrate groups, thus rendering the pond less attractive for flamingoes. However, since the response of flamingoes to eutrophication is not always negative, other factors can have been involved, e.g. the proliferation of filamentous algae, acting as a physical barrier against the access to prey living in the sediment or water column.

Sterna caspia was the indicator species of the winter 01/02 (it concentrated 100% of its abundance in that year). Again the lack of limnological data precludes interpreting the relationship between its presence and the environmental conditions of the pond, although it could also have been occasional, since Paracuellos *et al*. (2007) described its phenological status as such in SE Spain. It is a cosmopolitan species wintering in Subtropical Africa. In Europe, it is usually found in the shores of the Baltic Sea, in the coast of the Black Sea and in all the perimeter of the Caspian Sea. It lives also in Kazajstan and in regions further East (Cabo and Camacho, 1981).

In the winter 03/04 the indicator species was *Larus ridibundus*. Besides being a local breeder, it is the most abundant of those present in CP (maximum of 314 individuals in the winter 02/03), being considered as a typical generalist species. It was recorded continuously in the pond since the start of the surveys in 1999. According to the observations made by one of the data contributors (G.A. Ballesteros) in some years the low levels of the water table and the abundance of fish (*Mugilidae*) could favor the presence and feeding activity of this (and other) larids. Regarding the interactions with other species, a fact that can explain the decline of shorebirds in the last years of the survey period is the observation that in some wetlands, the numerical increase of the colonies of *Larus ridibundus* was responsible for a greater interspecific competition leading to the reduction of the populations of other waterbird guilds (Paterson, 1997).

Finally, *Larus michaellis* was the indicator species in the winter of 07/08. It is a species of wide geographical and ecological distribution, generalist in the choice of habitat and food and an opportunistic species with great trophic flexibility (Ramos *et al.*, 2009). At present, *Larus michaellis* colonizes not only islands, islets, cliffs and beaches (Purroy 1997), but also periurban areas and even cities due to the permanent availability of food in rubbish dumps (Bosch *et al.*, 1994, Burger and Gochfeld, 1996). Among the main biological effects of the expansion of *Larus michaellis* are its negative impact on other birds, especially through the displacement of endemic and endangered species (Swennen and Van de Meer, 1992) or through direct predation on many other (Oro *et al*., 2005). Moreover, their carrion feeding habits and their frequent use of rubbish dumps as feeding areas, increase their exposure to endoparasites and enhance their potential role as vector of infectious agents (Ramos *et al*., 2010).

Another species exclusive (100% of its abundance) in that same winter period was the Moorhen (*Galllinula chloropus*). Contrary to the majority of the species recorded, the Moorhen is a phytophagous species (Cramp and Simons, 1977), although the generalist and adaptable nature of this wide-ranging species precludes any attempt to draw conclusions from its occurrence in CP. In any case, its presence can be framed within the general succession process described by the change of the waterbird community.

As for those species exclusive (100% of their total abundance) in any other year, the winter of 1998/99 stands out with 5 species: *Ardea alba* and four shorebirds (*Haematopus ostralegus*, *Charadrius dubius*, *Vanellus vanellus* and *Calidris alba*). In the winter 01/02 four species are again found with this status (*Anas crecca* and the three shorebirds *Calidris ferruginea, Philomachus pugnax* and *Numenius arquata*). It can be seen that all except one of these species are shorebirds, which supports the dominance of specialists over generalists in the first period of the ornithological survey.

Returning to the summer community, in 2000 the indicator species for this period was *Pluvialis squaratola*, whose diet is composed, as in most shorebirds, of crustaceans, mollusks, worms and insects (Turpie, 1995). Like other shorebirds of similar size it forages in waters less tan 5 cm deep (Connors *et al.*, 1993), which further indicates that in the summer concerned, the water level was low and consequently there were large tracts of exposed shores available for this guild to feed. In the summer of 2002 the indicator role corresponded to *Tringa erythropus*, another medium-sized shorebird (Myers, 1980). It is in the summer 2004 when for the first time a gull species (*Larus ridibundus*) appears as indicator, and in the summer of 2006 this condition corresponded to *Larus melanocephalus,* a very scarce breeder of which two pairs nested in CP that same year. In the Regional Park the higher numbers of breeding pairs were recorded between 2004 and 2007. Given that, the presence of this gull in CP most probably resulted from the concurrence of the overall trend with suitable conditions in the pond area.

There were also species that concentrated all their abundance in any single summer: in 2001 two wading birds (*Bubulcus ibis* and *Ciconia ciconia*); in 2002 two gulls (*Larus audouinii* and *Larus fuscus*); and in 2008 a duck (*Anas platyrhynchos*) plus the Moorhen *Gallinula chloropus*. Among them, it should be highlighted the presence of Audouin's Gull (*Larus audouinii*). This gull has held one of its major breeding colonies in the nearby Isla Grossa (Oro, 1996), and 15 pairs were recorded in the Regional Park in 2007 (although not in CP). Its summer occurrence in CP leads in any case to a considerable increase in the value of the conservation indexes SPEC and LRAE. Its relative abundance depends largely on the presence and demography of *Larus michaellis* (Oro and Matínez Vilalta, 1994; Oro, 1996; Oro *et al.*, 1996; Martínez Abraín *et al.*, 1988). Despite being a larid, it can be categorized as specialist, due to its critical dependence on marine food resources.

As it is summarized in table 5, as a general rule, the species appearing as indicators (often being also exclusive) for years before 2004, belong to the "specialist type" (*Phoenicopterus roseus, Tringa erythropus, Pluvialis squatarola*). In this first period also acquires such status a tern (*Sterna caspia*), although it is an occasional visitor. Thereafter, all the indicator species identified are defined as generalists or opportunists, and belong to the same guild and family (*Laridae*): *Larus ridibundus*, *Larus melanocephalus* and at last, the most generalist one, *Larus michaellis*.

	95/99	99/00	00/01	01/02	02/03	03/04	04/05	05/06	06/07	07/08	08/09	09/10
Indicator species WIN		*Phoenicopterus roseus*		*Sterna caspia*		*Larus ridibundus*				*Larus michaellis*		
Indicator species SUM		*Pluvialis squaratola*		*Tringa erythropus*		*Larus ridibundus*		*Larus melanocephalus*				
Species with ABD 100% WIN	*Egretta alba* *Haematopus ostralegus* *Charadius dubius* *Vanellus vanellus* *Calidris alba*			*Anas crecca* *Calidris ferruginea* *Philomachus pugnax* *Numenius arquata* *Larus audouinii* *Sterna caspia* *Gelochelidon nilotica*			*Gallinago gallinago*			*Gallinula chlropus*		
Species with ABD 100% SUM			*Bubulcus ibis* *Ciconia ciconia*				*Larus audouinii* *Larus fuscus*			*Anas platyrrynchos* *Gallinula chlropus*		
Guild >%ABD WIN	SHBIRDS	SHBIRDS	SHBIRDS	GULLS	GULLS	GULLS	SHBIRDS	GULLS	DUCKS	GULLS	GULLS /DUCKS	
Guild >%ABD SUM	SHBIRDS	SHBIRDS	SHBIRDS	SHBIRDS	GULLS	GULLS	GULLS	GULLS	SHBIRDS	GULLS	GULLS	GULLS
Guild>RICH WIN	SHBIRDS	SHBIRDS	SHBIRDS	SHBIRDS	SHBIRDS	SHBIRDS	SHBIRDS	SHBIRDS	SHBIRDS	WADERS	SHBIRDS	SHBIRDS
Guild>RICH SUM	SHBIRDS	SHBIRDS	SHBIRDS	SHBIRDS	SHBIRDS	SHBIRDS	SHBIRDS	GULLS	SHBIRDS	SHBIRDS	SHBIRDS/ WADERS	SHBIRDS
Periods	**A: Dominance of Specialists**				**B: Transition period to Generalists**					**C. Dominance of Generalists**		

Table 5. Final summary of indicator species for years and temporal phases, derived from the analysis of total (summer and winter) census of waterbirds (ABD 100%: Total abundance of the species concentrated in that year; Guild >%ABD: Most abundant guild that year; Guild>RICH: Richest guild that year)

Shoreline units

The results regarding shoreline units are structured in four subsections. It starts with a more descriptive presentation of the spatial (between units) and temporal (between years) variation in biological indexes, followed by a gradient-based interpretation of the change in these same indexes. Then, the variation of conservation value indexes is presented and discussed, and finally an ordination-based analysis is made on the basis of similarity measures and indicator species recognition among the shoreline assemblages.

Spatial and temporal variations of biological indexes

In the lagoon shoreline, the changes in the indexes of abundance, species richness and diversity, and ornithological value of the waterbird community are assessed along shorter time scales, but the possibilities of spatial comparisons increase.

In winter, the Mar Menor community responds to internal structural and functional gradients, as the decreasing influence of the Mediterranean Sea into the lagoon -opposite to the degree of confinement- (Pérez-Ruzafa *et al.*, 2011), given its regime of transitional waters, the hydrodynamic conditions, its physical structure and geomorphology, and several external influential factors such as human activities (Robledano and Farinós, 2010). The most abundant species in the wintering community are coot, grebes, cormorants and gulls (mainly Yellow-legged gull *Larus michahellis*) as shown in Figure 6. The interannual variations in the abundance of some species like coot or cormorant are not clearly related with previous trends described for these species on a long term scale (Robledano and Farinós, 2010), however the dominant role of certain species in the wintering community (grebes, cormorants or Yellow-legged gull) remains as when examined at that long term temporal scale. The interannual variations detected (as the decreasing trend of coot and cormorants –degradation indicators-) could be related with the short-term variations in the intensity of eutrophication agents affecting the CL in recent decades.

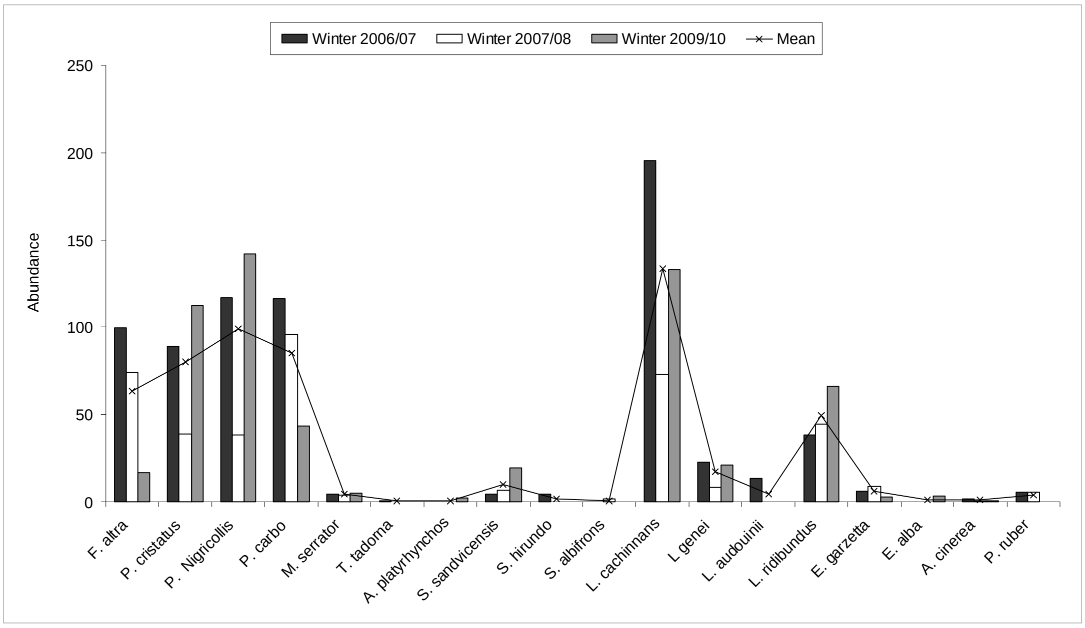


Figure 6. Total wintering abundance by species.

Differences in total abundance between the two main basins of the lagoon (northern and southern basin) were also detected (Figure 7). The southern basin, which includes the most natural areas (saline steppe, saltmarshes and reedbeds), displayed higher abundance values than the northern one (more anthropized).

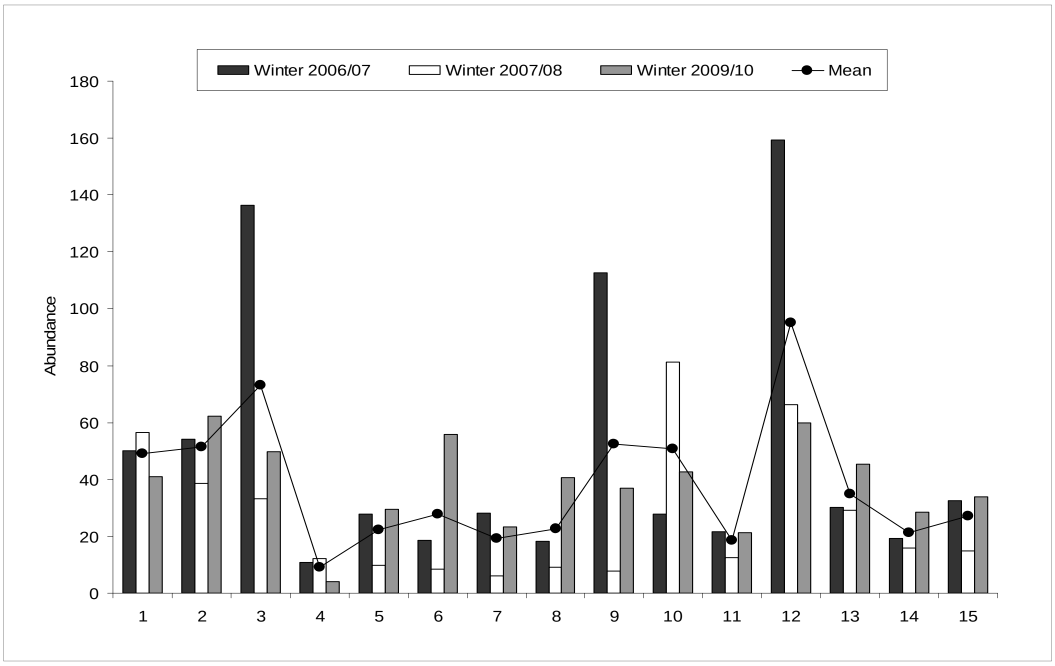


Figure 7. Total wintering abundance (per winter) in each sampling unit.

Furthermore, waterbird abundance differed significantly between bands, depending on trophic requirements and foraging strategies of different species (linked to the depth of the water column), or to the species-specific tolerance to human disturbance.

Comparing the present results with trends in wintering waterbirds recorded since the 70's, it is noted that Red-breasted merganser *Mergus serrator* maintains its gradual decline (Farinós and Robledano, 2010; Robledano *et al.*, 2011). This species, with a well-defined long-term negative response to the trophic enrichment of the lagoon, remains relict to oligrotrophic areas, in particular, clear and exposed littoral waters close to the Regional Park of San Pedro del Pinatar which, as a secondary wetland associated to the lagoon (from which in fact it originated), could be playing a shelter role for certain species (Taft and Haig, 2006). Conversely, grebes and cormorants display high abundances (although there is a gradual decline in the latter).

Great crested grebe *Podiceps cristatus*, Black-necked grebe *Podiceps nigricollis* and Great cormorant *Phalacrocorax carbo* are concentrated mainly in southern sectors with an intermediate level of eutrophication (related to an indirect influence of nutrient discharges through ephemeral channel outlets, probably due to their internal mobilization in the lagoon). As a result, the indicator role of these species, signalling moderate stages of eutrophication (Robledano *et al.*, 2011) is maintained.

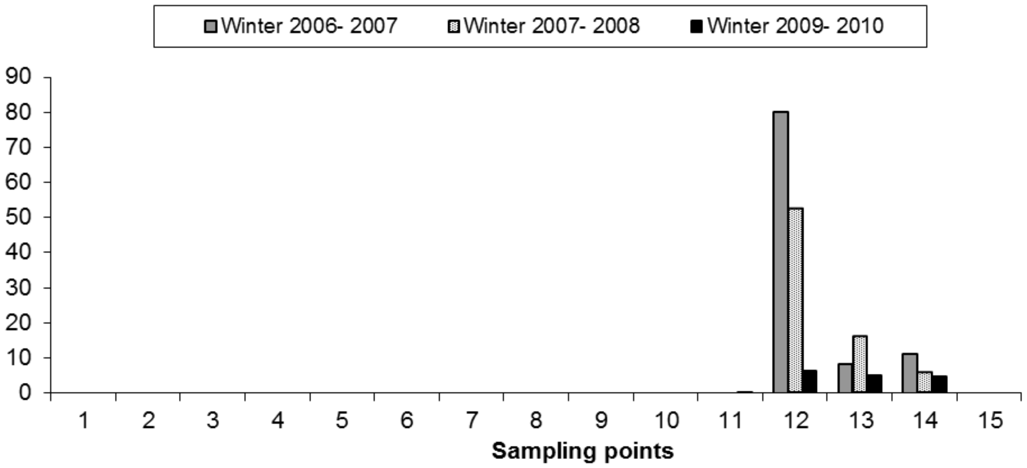


Figure 8. Distribution of *Fulica atra* among sampling units.

The most obvious response to eutrophication has been the increasing trend of wintering Common coot *Fulica atra* in recent years, linked exclusively to sectors with direct discharge of nutrients (Robledano *et al.*, 2011). Although its site preference remains the same in the present study (Figure 8), this recent period shows a gradual decline in coot numbers, which could indicate a slight setback in the process of eutrophication of the lagoon. Within the general state of incipient eutrophication, the complex interactions among biological compartments (phytoplankton, macrophytes, jellyfish) and the annual changes in wastewater inputs, agricultural drainage inflows, together with the climatic variability, can contribute to these short-term responses in waterbird numbers. However, it is feasible that the negative trends shown by coot in other nearby regions (Robledano *et al.*, 2011) have been finally mirrored by the populations wintering in the Mar Menor CL.

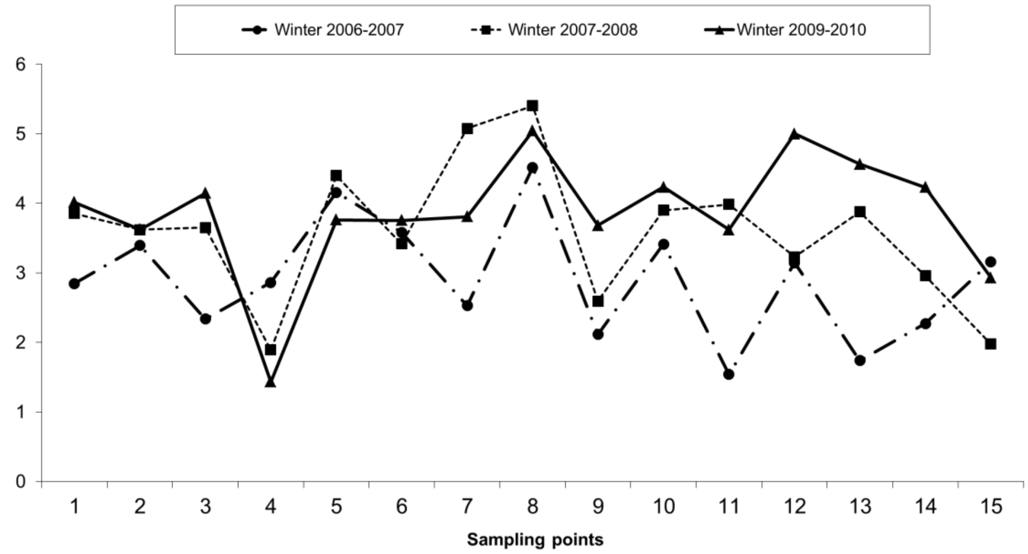


Figure 9. Variation of the Shannon-Wiener diversity index between years and sampling units in winter.

On the other hand, the high abundance of Yellow-legged gull with respect to other *Laridae* could be explained by its eclectic and opportunistic behavior, and the probable competitive disadvantage of other species of gulls, allowing this expansive species to thrive in the increasingly disturbed lagoon environment.

Regarding species richness and the Shannon diversity index, southern sampling units -in general with more natural conditions- present the maximum values (Figure 9).

These units receive the indirect influence of nutrient discharges, resulting in an intermediate eutrophication stage. Higher community diversity is a characteristic feature of waterbird assemblages of moderately eutrophic waters (Fjeldså, 2004; Rönkä *et al.*, 2005; MacDonald, 2006).

In addition to the higher productivity of their waters, the lower intensity of human activities in the units of the southern basin (Farinós and Robledano, 2010), and the species-specific tolerances to human disturbance (Burton, 2007) may be important factors that help explain these peak values of diversity and richness.

Differences in diversity and richness values were also detected between bands, since different species of waterbirds exploit the lagoon’s littoral waters at different distances from the shoreline, according to their morphotype, feeding strategy or specific sensitivity to human presence. No differences were detected between basins.

In summer the response of the waterbird community to internal and external gradients is mediated by the increase (with respect to the winter season) of water-based human activities, and in general of stressing factors affecting the lagoon (Farinós and Robledano, 2010). As in winter, the total summer abundance differs significantly from the south to the north basin, higher abundance values being concentrated in the southern division (with more natural features). In addition, significant differences were also detected between bands according to the same features commented in winter (trophic requirements and foraging strategies).

The summer community (Figures 10-13) is dominated by *Laridae* and allies. The Yellow-legged gull is the most abundant species in summer, followed by Little tern *Sterna albifrons* and Common tern *Sterna hirundo*. However, except for the two species of terns mentioned, all species have lower abundances than in the wintering season. It is necessary to explain the occurrence in the summer dataset of some essentially wintering species (like coot or grebes). This is due to the fact that just two broadly defined phenological divisions were considered (winter and summer). Therefore, these records come from the first transitional months included in the summer period. In any case, this differentiation of just two seasonal periods is consistent with the dynamics of human activities in the study area, whose benign climatic conditions allow typical ‘summer’ human activities (and the impacts derived from them) to take place along the six months of our *aestival* period (April to September).

Significant differences were found between bands in both richness and diversity, according to the features mentioned above. As in winter, no differences were found between basins for these two indexes.

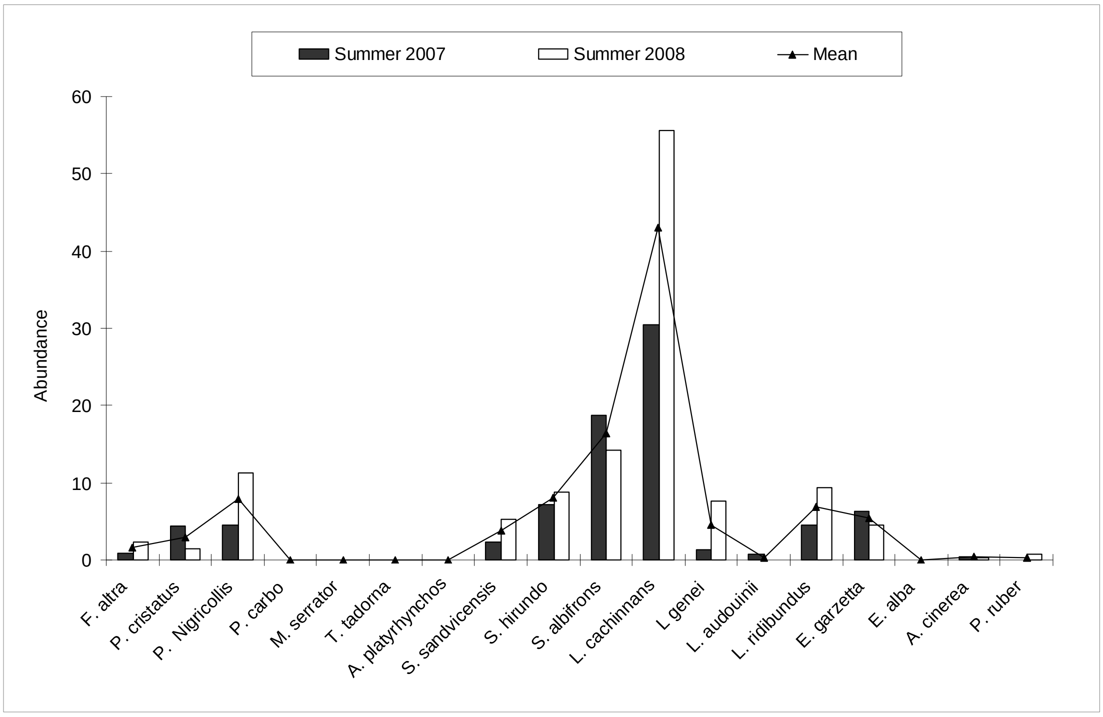


Figure 10. Total summer abundance by species.

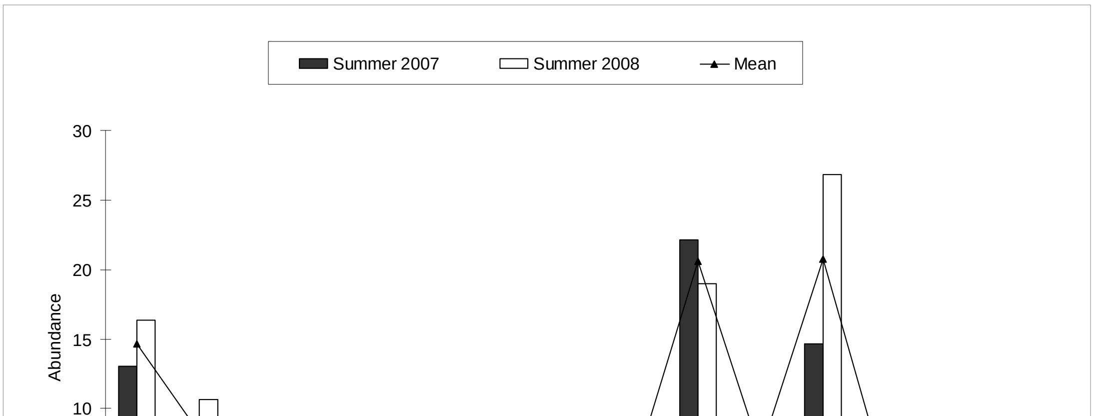


Figure 11. Total summer abundance in each sampling unit.

Is noteworthy that the highest mean values of diversity (Figure 14) were recorded in units 1 and 2 (protected natural area and main channel of communication with the Mediterranean, respectively) and unit 12 (mouth of the main channelized watercourse discharging into the CL).

Such peak values were expectable in these areas since they represent scenarios favoring biological diversity, but owing to different causes: around the protected natural area, due to its shelter role (especially in summer when human disturbance increases in all the CL); in the mouth of the main discharge channel, due to the eutrophication effect that boosts biological productivity (and hence a temporary peak of bird diversity); and in the main communication with the Mediterranean, due to the increase of resources characteristic of lagoon-sea transitions, and to the diversifying effect of such areas on waterbird communities (sharing species from both ecosystem types).

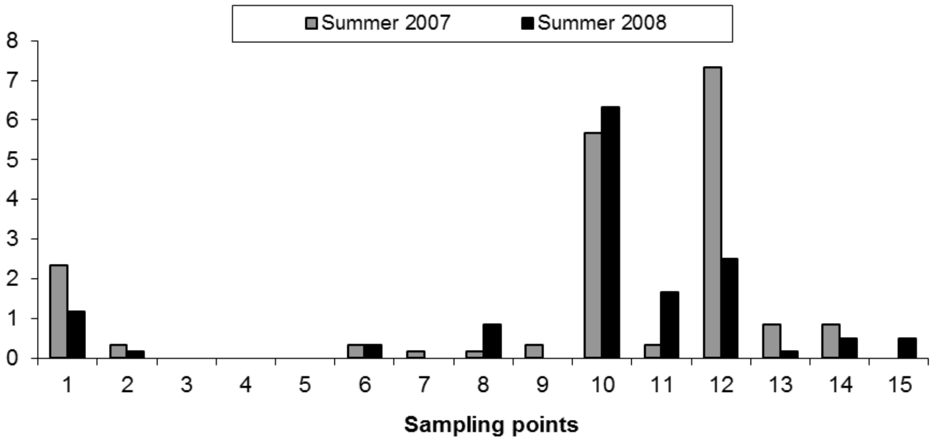


Figure 12. Abundance of Little tern *Sterna albifrons* in each sampling unit.

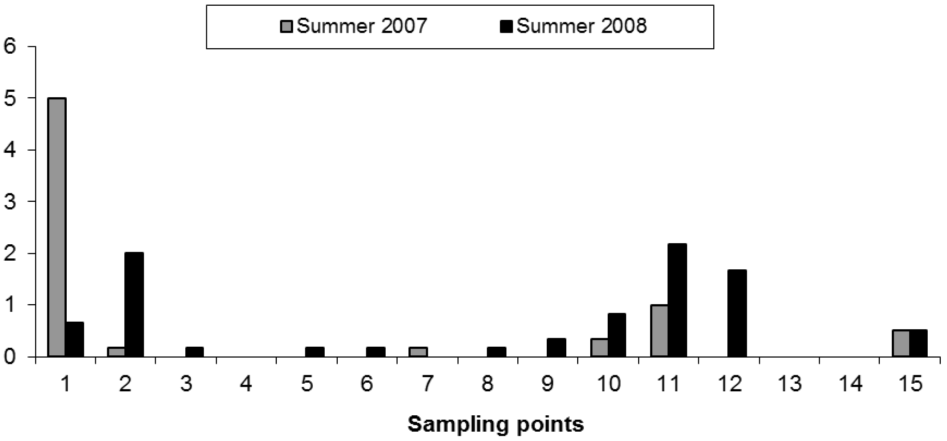


Figure 13. Abundance of Common Tern *Sterna hirundo* in each sampling unit.

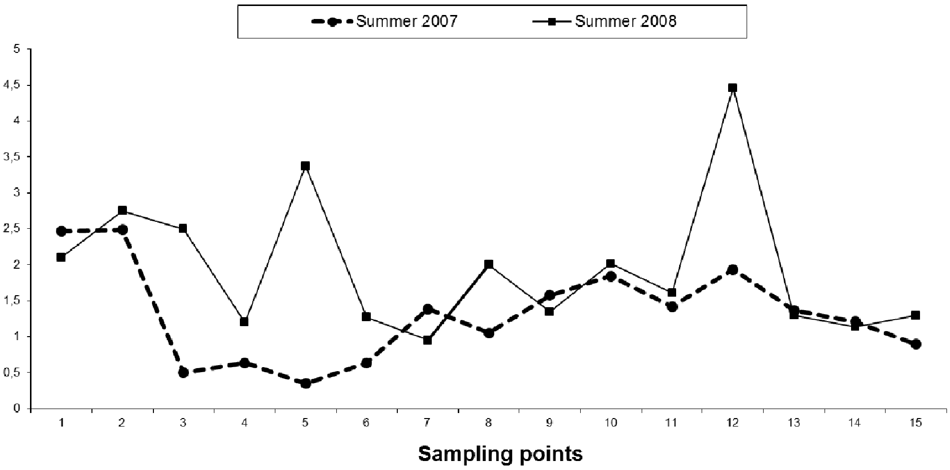


Figure 14. Shannon-Wiener diversity change between years and sampling units in summer

Main spatial and temporal gradients in biological indexes

After analyzing the temporal and spatial variation of biological indexes, it can be concluded that different temporal and spatial gradients are superimposed and affect the structuring of the waterbird community at different scales. First, a gradient of seasonal change in diversity and total abundance is found, with higher values in winter and lower in summer, related to the dominant phenology of the species recorded and to the intensity of human activities, respectively. This overall drop in diversity and abundance from winter to summer is a characteristic of southern coastal wetlands that receive an influx of wintering waterbirds, becoming refuges for some key species (Scott and Rose, 1996; Erwin, 1996). On the other hand, several overlapping spatial gradients are detected at different scales and both in winter and summer. There is a gradient of increasing abundance from the north to the south part of the basin, probably related to competition (intra and inter-specific) and to differences in the quantity and availability of food resources.

There are also typical variations in abundance and diversity between bands, which are related to the morphotype and foraging strategies of the species involved. Finally, the variation in abundance and diversity among units can be explained on the basis of habitat quality gradients, occurring again in both winter and summer. These gradients summarize the different affinity of the species with the various types of lagoon shoreline, depending on their habitat structure and land-use scenario, interacting with the waterbirds' specific tolerances to human disturbances, since each unit represents a particular human impact landscape.

Spatial and temporal variations of conservation value indexes

Unit 1 (belonging to a protected natural area) has the higher overall conservation value (considering both indexes) (Figures 15-16). Regarding the spatial gradient of conservation value within the CL, both indexes increase (both in summer and winter) in natural areas with different conditions: i) littoral waters suffering direct or indirect influence (due to internal mobilization) of agricultural drainages; ii) presence of other associated wetlands (probably offering alternative food resources or shelter), and iii) transitional areas that enhance biological diversity, like channels communicating the CL with the Mediterranean sea). The high conservation values attained by shoreline areas close to nutrient discharge points (and thus moderately enriched), seems related to the initial positive effect that the eutrophication process has on waterbird species richness and numerical abundance (Fuller, 1982).

With regard to the variation of LRAE index, it clearly shows a decrease in the conservation value of the waterbird community from winter to summer, since wintering species tend to have a higher protection status. The presence of Flamingo *Phoenicopterus roseus*, Slender-billed Gull *Larus genei* or Sandwich tern *Sterna sandvicensis* as wintering species in the CL, certainly contributes to the higher winter scores of the conservation inexes, while the summer community hosts more generalist and widely distributed species. At the beginning of the summer season there is a significant decrease in the LRAE index in almost all the CL (except for an increase in unit 10), related to the intensification of human activities (especially water-based ones).

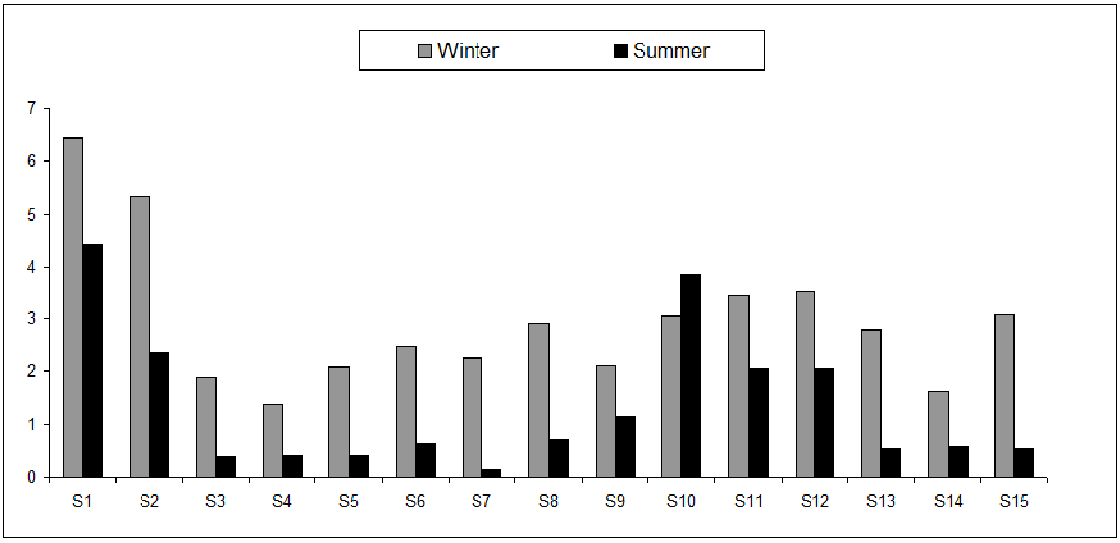


Figure 15. Mean values of the LRAE index in each sampling unit in both seasons. Winter values are means of 06/07, 07/08 and 09/10 data; summer values average 2007 and 2008 data.

In fact, it should be noted that unit 10 is one of the few areas remaining with a similar intensity of human activities (low level of recreational use in the adjacent land) in both seasons, which would confer it a shelter role in summer, when anthropogenic stress increases significantly in nearby areas.

With respect to seasonal variations in the index based on SPEC categories, again a decreasing trend occurs from winter to summer, also related to the higher protection status of many wintering species. It is noteworthy the remarkable increase in the conservation value of the community around unit 10 (also observed in the LRAE index, although less pronounced), for the reasons discussed above.

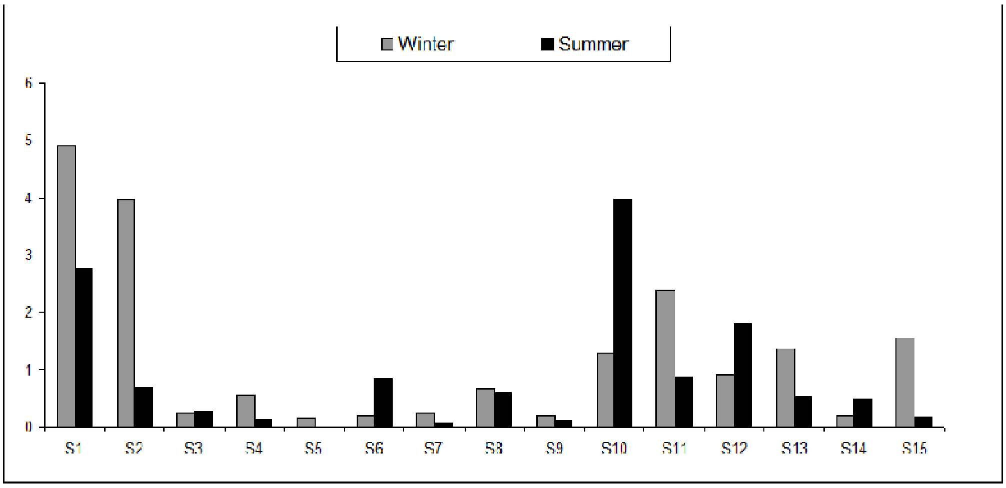


Figure 16. Mean values of the SPEC index in each sampling unit in both seasons. Winter values are means of 06/07, 07/08 and 09/10 data; summer values average 2007 and 2008 data.

Moreover, a similar increase in the conservation value of the summer community was observed in unit 6 (near to a protected area) and in units 12 and 14 (natural areas influenced by agricultural drainages). In addition to the greater trophic offer expected in these sectors experiencing moderate eutrophication, due their shared natural features (and sometimes proximity to protected areas), they could provide shelter for the summering species of higher protection status.

In short, it becomes evident that the winter community reaches overall greater conservation value, and that the higher conservation scores are also concentrated in natural or semi-natural areas, affected by various factors that can promote diversity and consequently improve conservation value (through the increase in species richness).

Similarity and indicator value analyses

MDS analyses were based on different grouping factors: band (B1 to B4), basin (North vs South), and LRAE and SPEC categories (sampling units were grouped according to rank values for each index). Figure 17 shows how there is greater cohesion in the winter community than in the summer one (Stress = 0.17). This ordination confirms the existence of two distinct communities: wintering and summer (breeding).

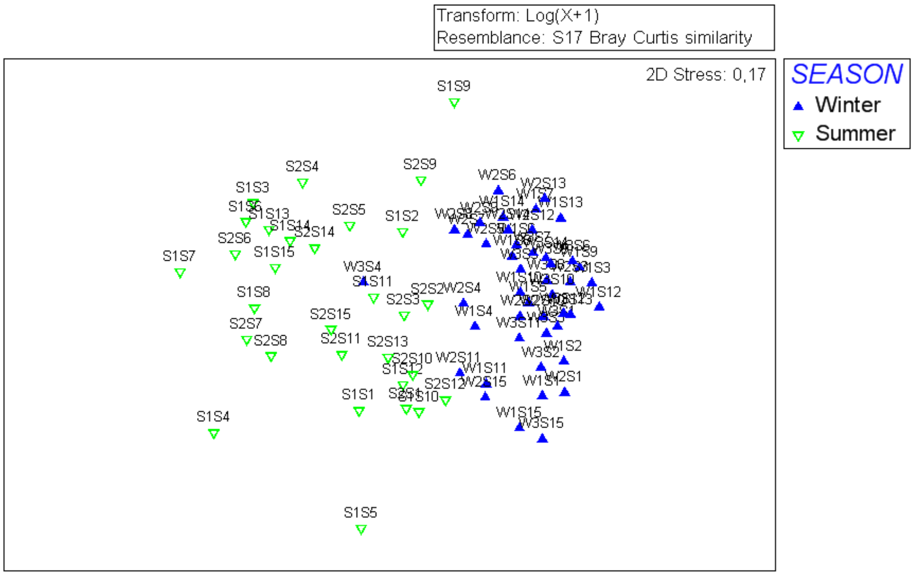


Figure 17. Multidimensional Scaling (MDS) differentiating samples on the basis of “season” as grouping factor.

With respect to the results of the analysis of winter data grouped by band, figure 18 shows how Band 1 is separated clearly from the other three (Stress = 0.18). This demonstrates that conditions of nearshore habitats determine a markedly different community composition in Band 1 (which attracts typical shallow water, wading and non-diving species).

Regarding the “basin” factor (Figure 19), it also emerged a grouping of units differentiating the northern from the southern one (Stress = 0.14). The southern basin appears as more cohesive (at least in waterbird community composition), probably related to the habitat characteristics shared by southern units. Also remarkable is the fact that the northern basin’s units are separated into the more eutrophic (S12 to S14) and the more oligotrophic ones (S1, S2 and S15).

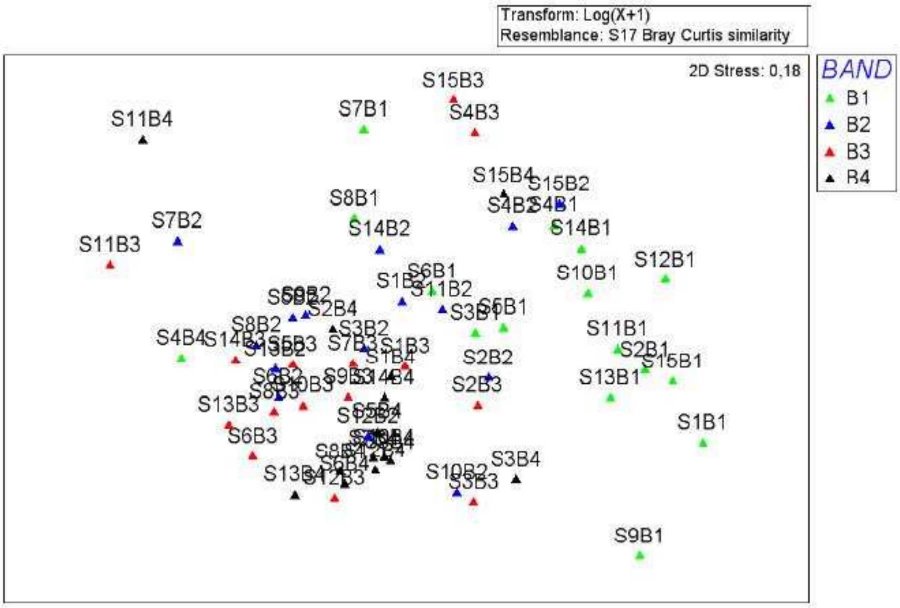


Figure 18. MDS differentiating winter samples on the basis of “band” as grouping factor.

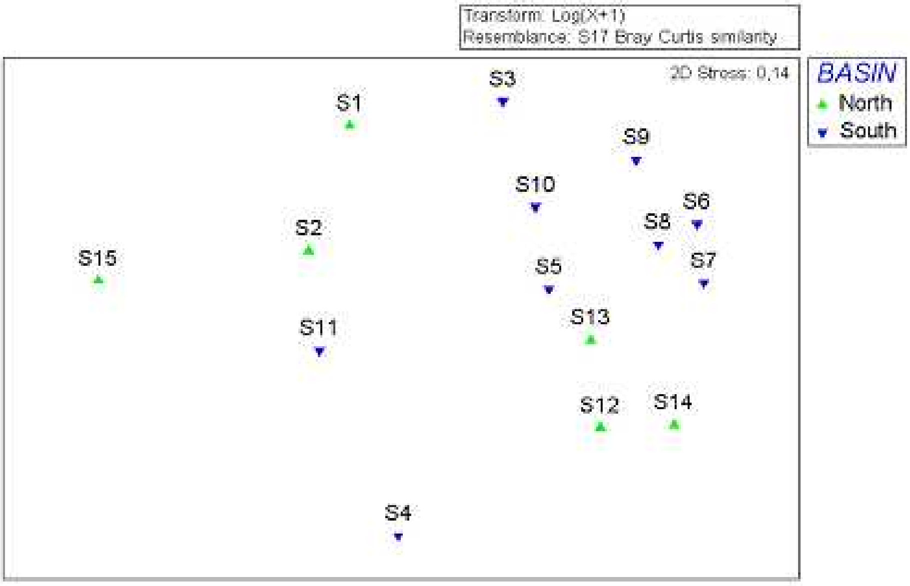


Figure 19. MDS differentiating winter samples on the basis of “basin” (North vs South) as grouping factor.

Furthermore, with respect to conservation indexes in winter (Figures 20-21), both display similar results: units with higher conservation values are grouped relatively homogeneously (Stress = 0.19). These are characterized by the presence of two species with high conservation value, Audouin's Gull and Flamingo.

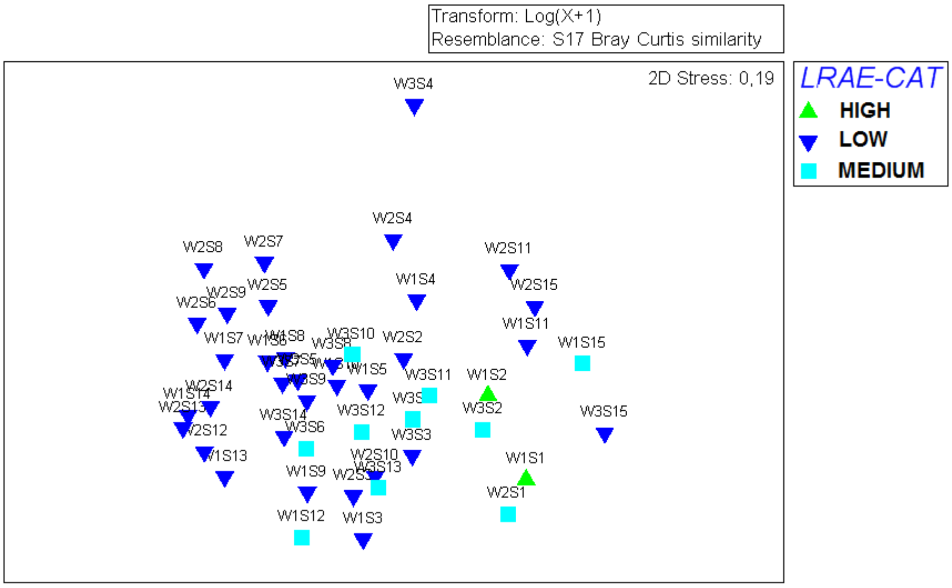


Figure 20. Multidimensional Scaling differentiating winter samples on the basis of "LRAE categories" as grouping factor

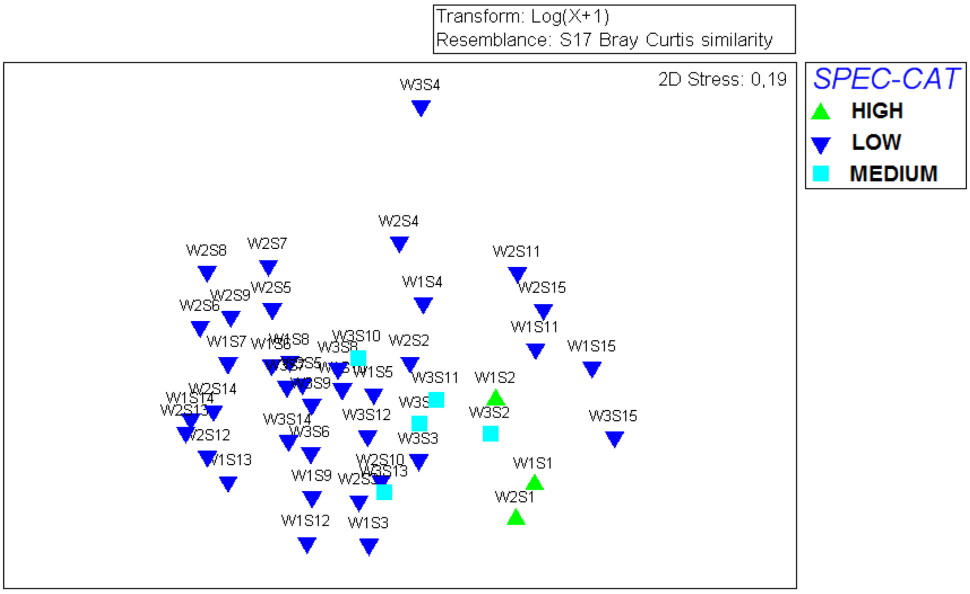


Figure 21. MDS differentiating winter samples on the basis of "SPEC categories" as grouping factor.

Among common features displayed by these units stand out the closeness to a natural protected area, the marine influence due to the proximity to the communication with the sea (Estacio channel), and the oligotrophic nature of their waters.

However, in both cases (LRAE and SPEC) the stress value of the ordination diagram is 0.19, very close but under the recommended limit of 0.2 considered acceptable in a multidimensional scaling analysis (Clarke, 1993).

Regarding summer data, the same grouping factors were tested and similar results were obtained than in winter. Band 1 is clearly separated from the remaining ones (Figure 22), revealing a summer community linked to shoreline habitats and other community linked to the littoral water column.

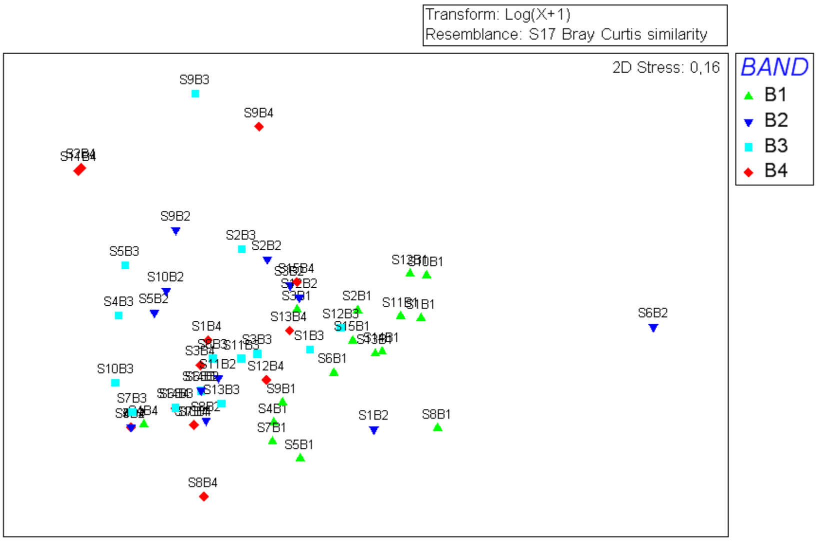


Figure 22. Multidimensional Scaling differentiating summer samples on the basis of "band" as grouping factor.

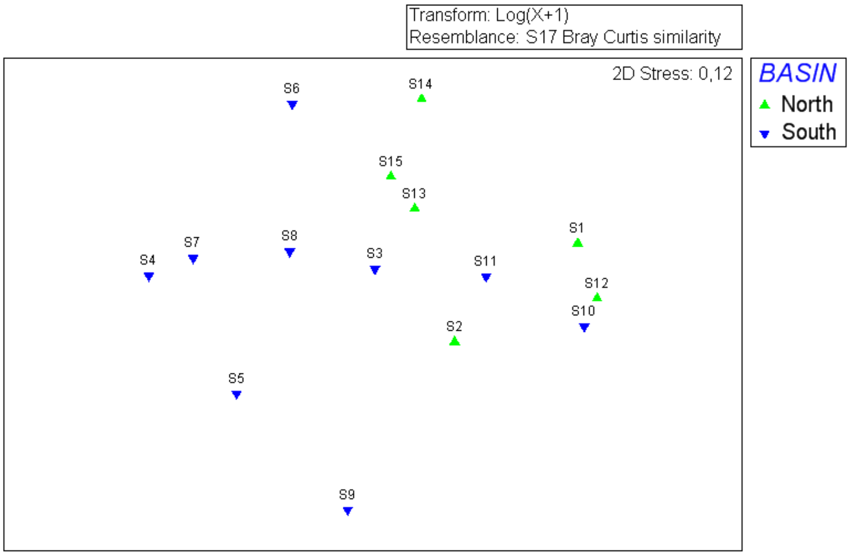


Figure 23. Multidimensional Scaling differentiating summer samples on the basis of "basin" as grouping factor.

Furthermore, the two basins are also separated in the ordination diagram (Figure 23) in a similar way to winter (Stress=0.12).

Finally, regarding conservation indexes, there is an apparent separation between units with a high or medium-to-high conservation value and units with lower conservation value, but the stress value (0.21) in both ordinations requires a cautious interpretation of the results. For that same reason, the ordination plots are not included.

Units of lower conservation value are characterized by a high degree of disturbance in summer, due to activities like bathing, boat traffic and other recreational uses. The units with higher conservation value are characterized by the presence of natural habitats adjacent to the shore, with minimal (or absent) representation of the above mentioned water-based activities.

After identifying groups of sampling units based on the factors studied (basin, band and SPEC and LRAE conservation indexes), "indicator" or representative species, based on the relative abundance of each group of units defined by MDS analyses, were identified trough subsequent IndVal analysis (Table 6). Significance level was set at p <0.05, although in certain specific cases, marginally significant results will be presented and discussed.

Factor	**Classes**	**Winter indicator species (p-value)**	**Summer indicator species (p-value)**
BAND	B1	*Fulica atra (0.03); Sterna sandvicensis (0.02); Larus genei (0.001): Larus ridibundus (0.001); Egretta garzetta (0.001); Ardea alba (0.012); Ardea cinerea (0.001)*	*Sterna sandvicensis (0.001); Sterna hirundo (0.001); Sterna albifrons (0.001); Larus michahellis (0.005); Larus ridibundus (0.001); Egretta garzetta (0.001)*
	B4	*Podiceps nigricollis (0.003); Podiceps cristatus (0.057)*	-
SPEC	High value	*Phoenicopterus roseus (0.002); Larus audouinii (0.067)*	-
	Medium value	*Sterna sandvicensis (0.036)*	-
LRAE	High value	*Sterna hirundo (0.038); Larus audouinii (0.047)*	-
	Medium value	*Egretta garzetta (0.069)*	-

Table 6. Indicator species in winter and summer for different factor classes; significance values based on Monte Carlo Test

In winter, for “band” factor, Coot, Sandwich tern, Slender-billed Gull, Little Egret, Great Egret and Grey Heron were identified as indicators of the first band. Coot, a phytophagous species, restricts its activity to the first few meters of littoral waters, while herons and egrets are wading species and thus feed in shallow water. On the other hand, Black-necked grebe stands as indicator species of the fourth band (relatively deep water). This expert diver and primarily piscivorous species, although with greater flexibility and opportunistic character than other members of the same guild (Fjeldså, 2004), can efficiently exploit this habitat. With respect to the factor "basin", no indicator species were detected.

In summer, Sandwich Tern, Common Tern, Little Tern, Yellow-legged Gull, Black-headed Gull, Little Egret and Grey Heron were identified as indicators of the first band. In the case of terns, this dependence on the first 100 m of water may be due to the specific requirements regarding the maximum depths they can exploit in plunge-diving. But it could also be related to the use of buoys delineating marine restrictions, as perchs or resting places. Finally, the indicator role of Yellow-legged Gull may be due primarily to the fact that with the decline of diversity in summer and the consequent temporary vacancy of certain ecological niches, this species takes over these empty niches, being more evenly distributed across the whole lagoon. The representativeness of the Black-headed Gull may be related to its wide range of diet (although still narrower than that of the Yellow-legged Gull), which also allows it to exploit several empty niches in summer. Neither indicator species were detected with respect to factor “basin”

In winter, regarding the SPEC conservation index, indicator species were detected for higher and medium categories. The indicator species of units with high conservation value were Flamingo and Audouin's Gull (although with marginal significance), present in more oligotrophic units, communicated with the Mediterranean sea and close to a natural protected area. The Sandwich Tern emerges as the indicator species of units with medium conservation value. This species is associated with maritime delineation lines of buoys (functional perches), located near the shoreline and being, a priori, the common element among the units with such medium category. There is probably some other common element to these areas which would help to explain the representative presence of Sandwich Tern, however it could not be detected at this level of environmental characterization of the units.

The analysis based on the winter value of the LRAE index, show Common Tern and Audouin´s Gull as indicator species of the maximum conservation category. These species were present in sampling units with overall high or medium conservation value. At least for the Audouin´s Gull, the reasons for this association are probably the same described for the SPEC index. It must also be highlighted the indicator role of Little Egret (with marginal significance) in medium conservation value areas. In summer, since there were not clear groups of units with respect to LRAE and SPEC conservation indexes, no Indicator Value analysis was performed to detect representative species of groupings based on their ranks.

Synthesis and potential application to wetland management and restoration

In the Mediterranean, CLs experience various influences that compromise the maintenance of their ecological and economic values. The effects of such influences are evident at different spatial and temporal scales, and should be tackled with cost-effective mitigation measures. At the same time, monitoring and alert systems are needed to avoid undesirable changes going beyond irreversible thresholds. Ideally these surveillance methods should be easy to implement and able to integrate a wide range of processes and impacts. While land, air or water-based multiparametric sensors can gather large datasets from multiple variables that need further integration and interpretation, bioindicators have the advantage to provide by themselves an integrative response to a variety of stressors (Kushlan, 1993; Adamus, 1996; Amat and Green, 2009).

In the absence of regular limnological sampling schemes, the changes of the waterbird communities can be used to infer broad environmental changes with ecological and economic significance. Their monitoring can inform management and trigger adaptive responses to the benefit of the whole wetland ecosystem, particularly in artificial or heavily stressed sites.

The results of the two studies carried out in the Mar Menor CL complex (described in this chapter), show the multiple physical scales and time spans over which waterbird communities change in response to natural and anthropogenic influences. Numerical and distributional changes can be combined in several community descriptors (species richness, diversity, guild structure) and indexes (conservation value), which provide information on spatial differences in quality and on temporal changes in ecological integrity. These parameters and indexes, as well as the occurrence of indicator species, can be used as signals of environmental change relevant for habitat management and biodiversity conservation.

Regarding the objective of assessing the utility of waterbird monitoring as a tool for tracking ecological changes in CLs at different scales, the local-scale study (Coterillo Pond system) has shown how waterbird changes can be summarized in a few temporal phases along which the turnover of guilds and species indicate a transition from specialists to generalists (parallel to a deterioration of the ecological status of the site, due to the persistent input of untreated effluents from a fish farm). At the end of this process of ecosystem succession, following some rejuvenation triggered by water flushing through the wetland, conservation value indexes increase due to the appearance of new species of different families.

At an intermediate scale (CL shoreline), the spatial and temporal analysis of the ornithological descriptors and indexes studied indicated that winter and summer waterbird communities differed in their composition and structure, but also in their interannual variation. In winter, total abundance was similar troughouth the whole study period, the dominant species being those associated with increasingly eutrophic waters like grebes, cormorants and some phytophagous species. There are spatial variations at different scales, between bands –which determine species assemblages depending on the physical conditions affecting them-, between sampling points, and between the two main basins (North and South) of the Mar Menor CL. In the southern one, the units with higher waterbird abundance and diversity are located near the outlets of ephemeral watercourse channels, where nutrient loads from the watershed of the CL are discharged in a concentrated manner.

Moreover, although the north basin supports various disturbance agents (harbours, airport), several waterbirds with high conservation value are still present, being species more related to open and oligotrophic waters, and apparently also highly dependent on the shelter provided by the nearby natural protected areas (adjacent wetlands).

The summer community shows inter-annual variations in abundance, richness and value of the conservation indexes. Abundance seems more determined in this case by the degree of naturalness of the sectors, than by trophic conditions. Differences between north and south basin are more diffuse, and there seems to be a more homogeneous distribution of the waterbird community, probably also related with an equally uniform distribution of human disturbance intensity throughout the Mar Menor

CL, and to the need of breeding species to forage closer to their respective nesting places.

The sectors with higher conservation value in both seasons, according to the two indexes computed (LRAE and SPEC), coincide in some features, i.e. natural habitat and landscape characteristics, and location away from urban centres, allowing them to host relevant species (regarding their threat degree and protection status).

But these features are lacking -and thus do not explain their conservation value- in other sectors with also high scores in these same indexes.

Regarding the second objective of evaluating the performance of different community descriptors and indexes as response indicators of ecological changes in CLs, it seems that monitoring schemes can benefit from the capacity of waterbirds to assess the implementation and effectiveness of management and mitigation measures (e.g. in relation to physico-chemical influences, changes in the landscape, in the intensity of land use, or in the degree of recreational pressure). Even so, part of the variability in waterbird community composition and structure remains unexplained, at least by the features used to define and characterize the ecological integrity (naturalness, hydrological alteration, openness) of our sampling units.

On the other hand, our third objective of integrating the results of waterbird monitoring programmes, run at different spatial and temporal scales within CL complexes, into a general evaluation system of their ecological integrity, depends critically on the availability of minimum financial resources to maintain a regular integrated survey of key biological (waterbirds, fish) and environmental variables (e.g. limnological conditions) in test sites like restored ponds and other man-made wetlands.

The last objective, devising a habitat restoration scheme based on the experience in waterbird monitoring of representative wetland types within the Mar Menor CL system, in order to maximize biodiversity and environmental functions (e.g. water purification) of the restored and created habitats, needs also a multi-scale spatial approach. The capacity of restored or newly created wetlands to buffer the effects of increased freshwater inputs and excess nutrients (from agriculture, urban areas, aquaculture facilities) can also be exploited to improve biodiversity at a local (site) and landscape (complex) scales.

At the local scale small wetlands can supply a wide range of ecosystem services (water purification, recreation, wildlife habitat). At the watershed scale, a network of artificial wetlands can exert (with respect to the whole CL complex) a buffering function which otherwise would be performed by natural wetlands or by the main CL basin (Alvarez-Rogel *et al*., 2011; Robledano *et al*., 2011), but in that case at the expense of their biological and landscape values. Our experience in ornithological monitoring and conservation assessment of both natural wetlands (Martínez *et al.*, 2005; Robledano *et al*., 2008; 2010; Farinós and Robledano, 2010; Robledano *et al*., 2011), and artificial or restored ones (present chapter), reveals that conservation gains in unmanaged areas can be rapidly offset if effective monitoring and consequent adaptive management are not implemented, and wetlands are simply used as passive "green filters".

In any case, indirect evidence from ornithological monitoring suggests that the lack of management and monitoring, or the performance of opportunistic, short-lived versions of these, not supported by carefully planned field-based research, inevitably leads to a loss of value under an unpredictable, often unstable or low quality ecological status.

Thus, to balance the provision of environmental services and the conservation of ecological values, a two-stage design is needed, in which a part of the restored or created wetlands is used for the processing of the water supply (e.g. nutrient enriched effluents), while another part receives the treated water and can be managed to enhance biodiversity. Each section is engineered in a way that increases the performance of its main function, selecting the vegetation type, morphometric characteristics and flow regime best adapted to it.

Such wetlands usually represent anomalies in the prevailing environmental context (e.g. freshwater habitats within a saline landscape), which allows biodiversity to be enhanced both at the local and landscape scale.

Recently, at the request of the regional environmental authority, a consultant team (with participation of the University of Murcia), developed a proposal for the treatment of agricultural effluents that discharge into the Mar Menor CL from the adjacent coastal plain through the wetlands of its western shore (Esteve *et al.*, 2010).

The proposal takes the form of a set of actions framed in the principles of ecological engineering (Marques *et al.*, 2003), combining actions at different scales. The set of actions were part of a dual and complementary approach of wetland re-creation and management:

1) First, the development of a Local Management System, applicable at a small spatial scale and focusing on the management of specific elements of the system (such as drainage canals or temporary ponds). At this level the main proposal was the creation of artificial wetland systems (functional units) composed by two ponds with different and well defined main functions: i) green filters, and ii) biotope recovery and increase of biodiversity.

2) Second, in a complementary way, the development of an Integrated Management System, applicable at a landscape scale. Basically, the proposal consisted in the joint management of the system of drainage canals, for concentrating and redirecting the water flows towards the functional units (local management system) that would perform the depuration role.

Once the combination of these two management systems was established as a basic wetland creation and management scheme, specific guidelines for the intervention in the area were set, in the sense of:

a) Increasing water retention times to allow better depuration process of the organic content (managing drainage channels and artificial wetlands)

b) Recovering saline steppe and saltmarsh biotopes and temporary water bodies associated to these habitats

c) Increasing biodiversity, through the restoration of native vegetation and the recreation of biotopes, favoring the structural heterogeneity in the functional units (ponds)

d) Controlling the expansion of reedbeds

e) Establishing scientific and research criteria when planning the monitoring and evaluation schemes of the restoration actions and their effects, both on the present biota and on the quality of the waters draining into the lagoon.

As an example of the types of actions proposed, figure 24 outlines two types of pond which would integrate an ideal two-stage wetland system for mitigating the impact of agricultural drainages channeled to the Mar Menor lagoon through the saline steppe wetlands of its western shore. We advocate the use of these artificial wetlands in the treatment of agricultural drainage and other effluents impairing the lagoon ecosystems, based on the recognition of their potential to mitigate such impacts, while increasing habitat for various species.

Monitoring waterbird communities present in these wetlands -and in the natural ecosystems for which they act as buffers-, will be one of the main tools for evaluating their effectiveness. Experience with the conversion of ancient lagoon-based sewage treatment systems into endangered waterfowl refuges (e.g.: http://www.lifemalvasiamurcia.es/pdf/ propuesta_zepa_ lagunas%20moreras.pdf), illustrate the potential for developing suitable waterbird habitats as by-products of water quality management systems in heavily modified landscapes.

Finally, table 7 summarizes the different space and time scales over which the waterbird-based indicators change in response to different environmental drivers, the type of response shown and the management options available for addressing the effects of these drivers, particularly when related to anthropogenic influences on CLs.

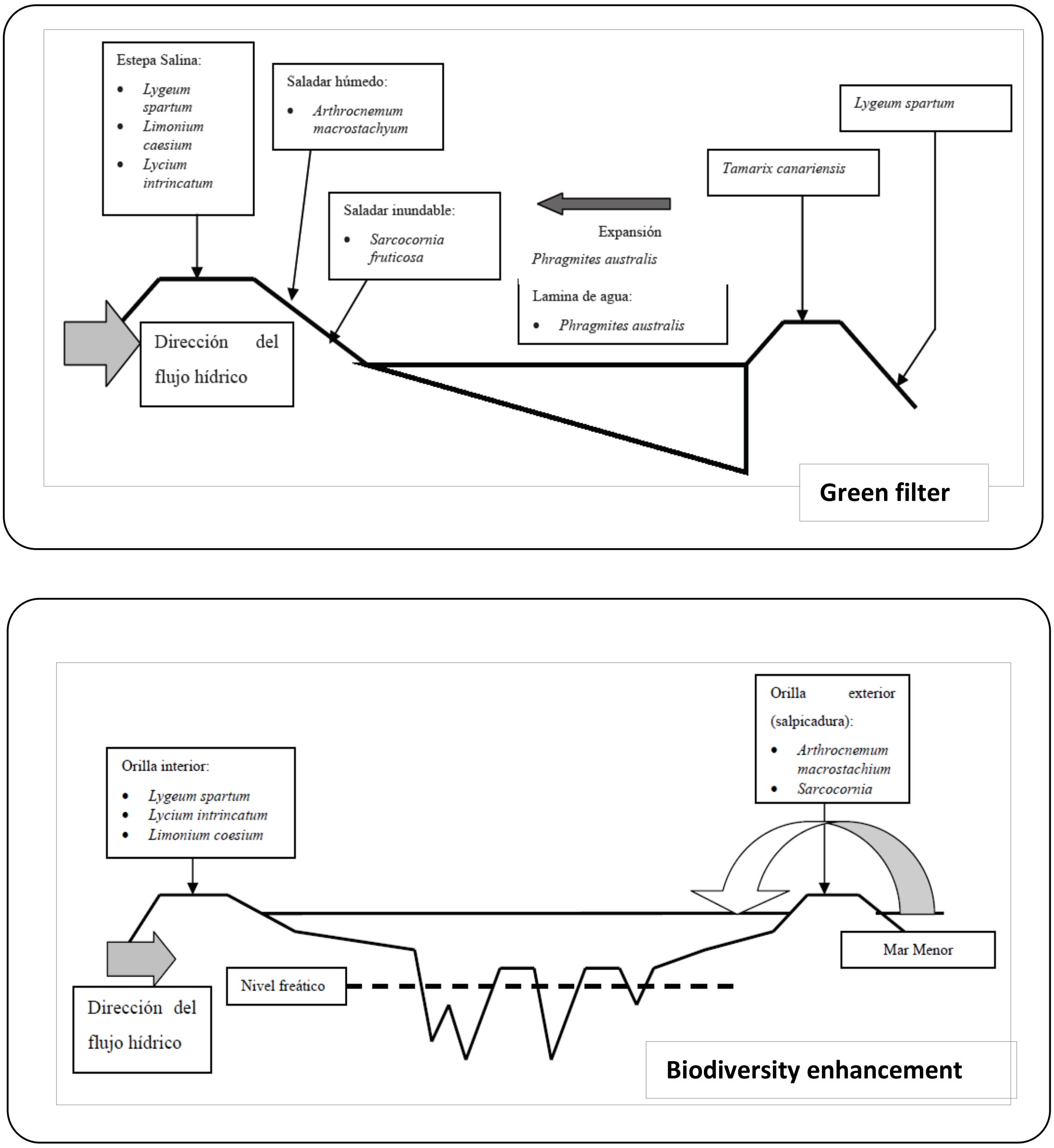


Figure 24. Schematic representation of the two types of artificial ponds which would integrate an ideal two-stage wetland system for mitigating the impact of agricultural drainages on the Mar Menor CL complex: on top “green filter” type (GF), and at the bottom “biodiversity enhancement” type (BE). On each, the arrow on the left side represents the direction of the water flow passing through the system. In GF, the narrower arrow represents the expected direction of expanding reeds (*Phragmites australis*). In BE, the right arrow represents the lagoon’s influence (splash), and the dashed line the phreatic water table (from Esteve *et al*., 2010).

Scales		Driving forces	Type of biological response	Management options
Space	Time			
Local	Short-term (seasonal)	Phenology Recreational pressure Habitat structure Seasonal course of activities, resource use and flows	Short-term and small scale numerical and distributional changes (band, season...)	Two-stage restored or created wetlands for addressing point-source impacts (green filter + habitat enhancement) Local restoration projects Routine management (water levels, recreational uses)
Intermediate (sampling sectors, internal gradients)	Mid-term (interannual)	Climate variability Reclamation projects Land use Recreational pressure	Changes at intermediate scales (among sectors/years, along spatial gradients)	Buffering systems of two-stage wetlands (to address diffuse impacts over medium-sized areas)
Whole CL complex	Long-term (decadal)	Climate change Socioeconomic trends Nutrient loading from agriculture and urban wastewater Agricultural drainage Landscape effects of land-use change Waterbird population trends at higher scales	Long-term habitat changes Noxious species blooms (microalgae, jellyfish) Waterbird species trends and community change (guild structure, diversity)	Networks of intercepting canals and two-stage wetland systems (buffering belts)

Table 7. Summary of spatial and temporal scales of waterbird community variation in response to impacts and other driving forces, and of management options available at each of these scales

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Chapter 4. Anthropogenic effects on habitat and landscape gradients influencing the littoral waterbird assemblage of a mediterranean coastal lagoon (Mar Menor, SE Spain)

Introduction

The need to understand how wildlife responds to the broad-ranging impacts of development is becoming increasingly important as human populations continue to grow and urbanization spreads around the globe (Crutzen, 2002). Integrated, quantitative expressions of anthropogenic stress over large geographic areas (e.g. watersheds and their associated aquatic ecosystems) can be valuable tools in environmental research and management, and a basis for the study of the response of bioindicator taxa to human activities (Brazner *et al.,* 2007; Danz *et al.,* 2007). The detection of potential bioindicator species, their monitoring and the ecological interpretation of their numerical responses provides also a useful management tool (Martínez-López *et al.*, 2014a). It allows relating ecological conditions to anthropogenic processes, thus helping in the development of cost-effective environmental indicators and in the implementation of remedial actions (Howe *et al.*, 2007). In regard to Europe’s aquatic ecosystems, the Water Framework Directive (WFD; European Comission, 2000), which came into force in December 2000, sets a target scenario which obligates to monitor the ecological status of surface waters, including coastal and transitional ones. The WFD stipulates that surface water bodies should achieve good ecological and chemical status by 2015 (Rowan *et al.*, 2006). Coastal lagoons (CLs hereafter), as semi-enclosed coastal systems (Newton *et al.,* 2014), and Mediterranean lagoons in particular, are especially suitable to test the responses of biota to human stressors, given the occurrence of intensive activities associated with a high density of human population around their shores (Viaroli *et al.,* 2005).

Waterbirds are recognized as useful ecological and environmental indicators. The role of waterbirds has been widely addressed both in general reviews (Gregory *et al.,* 2003; Stolen *et al.*, 2004) and in case studies (Paillison *et al.,* 2002; Palacio-Núñez *et al.*, 2007; Robledano *et al.,* 2008; 2011), being based on various avian properties. Many studies on the effects of human activities on waterbirds have been conducted in well-preserved habitats, often within nature reserves or areas managed for conservation (McKinney *et al.*, 2006). Wetlands close to high density urban areas, which also provide valuable habitats for waterbirds, may experience greater pressure from human activities through modifications of landscape characteristics (Cardoni *et al.*, 2011), such as watershed processes, that can result in severe structural or functional change, even without direct habitat alteration (McKinney *et al.*, 2006; Carreño *et al.*, 2008; Martínez-López *et al.*, 2014b; 2014c). In relation to waterbirds and aquatic environments, the scale at which structural or functional changes occur will determine different management scenarios. Immediate habitat-scale responses are probably associated with a positive or negative structural effect (vegetation cover, disturbing physical elements, etc.). Responses to landscape configuration in adjacent areas or in the whole catchment can also have a structural basis in some cases (Webb *et al.*, 2010; Pearse *et al.*, 2012), but rather a functional one, through the influence of processes like agricultural irrigation and fertilization or urban pollution (Martínez *et al.*, 2005; Velasco *et al.*, 2006; Pardo *et al.*, 2008; Cardoni *et al.*, 2011). Thus the combined analysis of biological responses on both habitat and landscape scales may be useful for the integrated management of wetland communities and their associated terrestrial landscapes in a context of environmental change (Fuller, 2012).

Traditionally used as evaluation criteria for nature protection at various geographical and administrative scales, waterbirds stand out as the main criterion - and for a long time, virtually the only available - for the designation of internationally important wetlands, especially under the Ramsar Convention. But quite often, once sites have been legally protected, waterbird numbers have become mere “success indicators” of management focused at enhancing bird populations and their habitats (Robledano and Farinós, 2010). Besides monitoring work, birds have received little scientific attention in CLs compared to research efforts directed to other taxa, ignoring their important role as top predators and their contribution to various ecological processes (Robledano and Farinós, 2010). Moreover, waterbird monitoring results are rarely integrated in the development of environmental policy and management, with the exception of Ramsar wetlands and Natura Network 2000 evaluation, and even in such cases, rarely beyond site designation (Robledano *et al.*, 2010). The exchange of knowledge between research on waterbirds and other lagoon biocoenoses (*e.g.* benthos) is also anecdotic (Robledano and Farinós, 2010). In line with this, birds are not included the WFD’s assessment of European waters. However, the knowledge of their response to anthropogenic pressures, can help to harmonize the WFD’s water-ecological quality targets with the biodiversity conservation objectives set by other EU Directives (such as the Birds and Habitats Directives). Such help can be especially useful in complex areas like Mediterranean

CLs and their associated wetlands, where many conservation and protection regulations and human activities overlap.

The particular case of the Mar Menor Coastal Lagoon (MMCL) summarizes all these issues. It is subject of a variety of environmental pressures (Pérez-Ruzafa *et al.*, 2002; Marín-Guirao *et al.*, 2005a; Álvarez-Rogel *et al.*, 2007; García-Pintado *et al.*, 2007; Pérez-Ruzafa *et al.*, 2007; Carreño *et al.*, 2008; García-Sánchez *et al.*, 2012). The waterbird community is one of its most important biological components. In fact, the area has been designated as SPA for its populations of Annex I species of the Birds Directive 2009/147/CEE, like Little egret *Egretta garzetta*, Black-winged *Himantopus himantopus* and Lesser Short-toed Lark *Calandrella rufescens* and it also hosts species which serve as indicators of the effects of agricultural intensification and urban development like grebes (*Podiceps nigricollis* and *Podiceps cristatus*) and Red-breasted Merganser *Mergus serrator* (Hernandez and Robledano 1997; Martínez *et al.*, 2005). The MMCL represents a key resource for its area of influence, to which it provides a variety of products (fisheries, salt) and services (transport, recreation and tourism), and from which it also receives a wide range of pressures (hydrological alterations, chemical inputs, structural modifications). Consequently it has been intensively studied in several ecological and environmental aspects (Pérez-Ruzafa *et al.*, 2002; 2004; Marín-Guirao *et al.*, 2005b; Lloret *et al.*, 2005; Perez Ruzafa *et al.*, 2005; Salas *et al.*, 2006; Velasco *et al.*, 2006; Pérez-Ruzafa *et al.*, 2012), including studies focusing on waterbirds and their relationship with long term environmental change and internal spatio-temporal gradients (Farinós and Robledano, 2010; Robledano and Farinós, 2010, Farinós *et al.*, in press). However, a critical part of the MMCL waterbird community lacks of specific studies, precisely the one occupying the most dynamic, productive and directly stressed sections: the nearshore habitats, whose dynamism and transitional habitat character enables a high diversity of ecological niches, thus bringing together a high diversity of waterbird species (divers, swimmers, waders, etc.). In other transitional and shallow coastal areas, a number of studies have related waterbird indices with environmental data gathered and integrated at different geographical scales (McKinney *et al.*, 2006; Taft and Haig, 2006; Danz *et al.*, 2007). This paper seeks to fill this gap in MMCL, looking for relationships between indices of waterbird use of the littoral sections and descriptors of human pressure in their areas of influence.

The objectives of this study were (1) to evaluate the spatial and temporal variation of waterbird-based indices in the nearshore habitats of the MMCL; (2) to identify indicator species of different types of nearshore habitats (in relation to land use and stress gradients at different spatial scales); (3) to describe the relationship between the environmental gradients and quantitative measures of waterbird indicator species and biological indices; and (4) to propose management strategies for the nearshore lagoon sections and their surrounding terrestrial landscape, which maximize ornithological value while preserving the ecological integrity of the MMCL ecosystem, as a contribution to the objectives of the European Union's Water, Birds and Habitats Directives.

Methods

Study area

The Mar Menor is a hypersaline coastal lagoon located in southeastern Spain (Fig. 1). With a surface of 135 km^2 and an average depth of 4 m (Esteve and Martínez, 2003), it is the largest coastal lagoon of the western Mediterranean. It is surrounded by ca. 600 km^2 of irrigated agricultural plain (2008 data) inside a total watershed area of 1.275 km^2 (Carreño *et al*., 2011) with dense touristic urban developments, and it is almost enclosed from the Mediterranean Sea by a sand bar (La Manga), also dominated by urban development. Since 1994, the Mar Menor is also designated as a Ramsar Site and since 2001 as a Bird SPA and a Specially Protected Area of Mediterranean Importance.

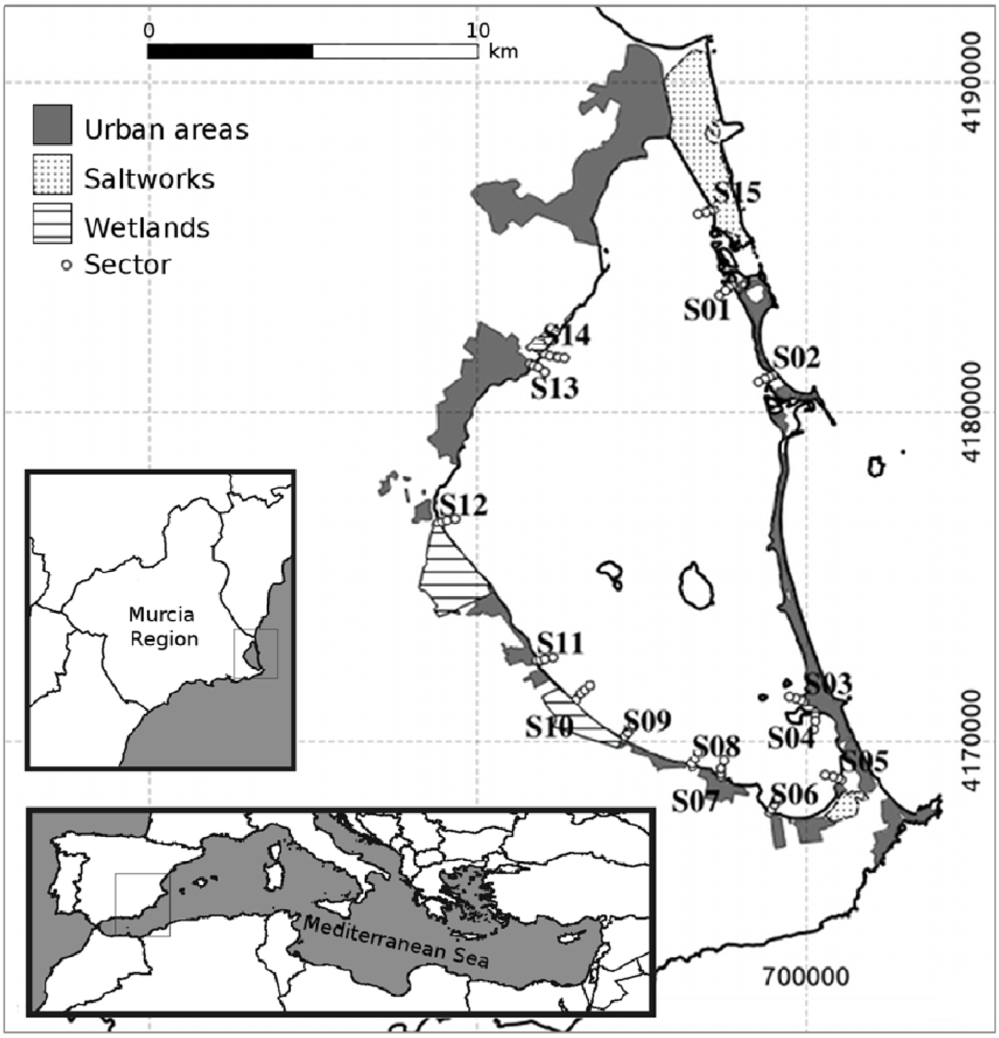

Figure 1. General map of the Mar Menor Lagoon showing the 15 bird sampling stations (=sectors), the urban areas and the main associated natural and semi-natural habitats: active saltpans (="salinas") and saline steppe areas and other phreatophytic formations (="criptohumedales").

The MMCL is surrounded by a number of wetlands historically modified by man for different purposes (mainly salt production). The coastline is fringed by patches of saltmarsh and salt steppes, formerly saltworks which have been abandoned during the 20th Century, interspersed with agricultural (traditional or intensive) and urban interfaces.

The Tajo-Segura river diversion, reaching the MMCL coastal plain in 1980s, changed its agricultural use from extensive dryland and traditional groundwater-fed to intensively irrigated crops. This caused significant hydrological changes (phreatic level rise, permanent agricultural drainages), and subsequently affected the structure and relative distribution of littoral habitats (Carreño *et al.*, 2008). Other major physical and hydrological changes had begun to occur in the lagoon water mass since 1970s due to the dredging of one of the channels linking it with the sea. This increased the marine influence starting a process of “mediterraneization” (Robledano *et al.,* 2010) that results in a more marked continental-oceanic gradient within the lagoon (Pérez-Ruzafa *et al.,* 1991)

Bird census

Monthly counts of waterbirds were conducted from October 2006 to October 2008 (two full annual cycles) and from October 2009 to March 2010 and October 2010 to March 2011 (two distinct wintering seasons) at 15 sampling stations distributed along the MMCL shoreline (Figure 1; see Farinós *et al.,* 2013). Stations were chosen so as to represent the main diversity of structural and functional characteristics of shoreline habitats in the lagoon. All waterbird species were censused in the sampling stations, except small shorebirds. The criterion was to include species able to exploit the widest range of the studied sections. This led to exclude small wading birds (e.g. *Calidris spp., Charadrius spp.*, etc), which due to their morphology are mainly restricted to micro-tidal habitats, and whose abundance in the MMCL is much lower than in nearby wetlands (saltpans, natural lagoon channels).

Following Robledano *et al*. (2008), the nearshore water section adjacent to each observation point was divided into four parallel bands at different distances from shore (B1=0-100 m, B2=100-250 m, B3=250-500 m, B4=500-1000 m), resulting in 60 sampling units (bands × sampling stations). The outer limit of the fourth band was set at 1000 m since birds could not be adequately identified beyond this limit. Each sampling station included 500 m of shoreline and was accesible by car. The same observer, trained in the estimate of distances, did all the censuses in order to minimise observer bias. Total abundance of each species was recorded in each unit during a 10 min observation period (per station) performed within the first 6 h of daylight. The starting point of the waterbird census was alternated between stations every month in order to minimise the effect of the time of the day on bird distribution.

Data analyses

Bird data

Based on previous studies in this area (Farinós and Robledano, 2010), and from general recommendations for waterbird monitoring (Van Roomen *et al.*, 2006), monthly bird counts were grouped into two seasons: summer (April to September) and winter (October to March). Although there is some overlap in the spring and autumn transition months, from an ecological perspective community organization is well suited to that temporal division, since there are two well-defined phenology-based waterbird communities (Hernández and Robledano, 1997).

Three waterbird-based indices were calculated for each sampling unit (station x band): Richness (R), Shannon-Wiener Diversity (H) and Total Bird Use (TBU; McKinney *et al.,* 2006). Their variation was tested with non parametric analyses of variance: Wilcoxon Test was used to analyze inter-seasonal (winter *vs.* summer) and inter-annual (between summers) differences (factor classes ≤ 2), and Kruskall-Wallis Test to analyze differences between winters (inter-annual), sampling stations (lagoon’s spatial heterogeneity) and bands (site heterogeneity) (factor classes > 2). Statistical significance was set at p = 0.05. When overall significant differences occurred, post-hoc paired comparisons were performed with the “pgirmess” package (http://giraudoux.pagesperso-orange.fr/). Finally, to assess the variation of indices in relation to the distance to shore, linear regressions were performed, with the parallel bands (1-4) at increasing distance from the shore (BAND) as dependent variable. All analyses were conducted with the freely distributed R software (R Development Core Team, 2007).

To analyze the spatial structure of the winter and summer community, multivariate methods of ordination and classification were used. After data transformation (log [x+1]), a Multi-Dimensional Scaling (MDS) ordination (Clarke, 1993) was performed with Primer 6 software (Clarke and Gorley, 2006) to analyze the effect of distance from the shore (BAND) on the waterbird community composition.

Definition of landscape gradients and selection of environmental predictors

To detect the main stress and land use gradients characterizing MMCL it was necessary to undertake a pre-analysis over two sets of variables (described in table 1), which were calculated with GRASS Geographical Information System (GRASS Development Team, 2011).

VARIABLE CODE	DESCRIPTION
Spatial and temporal factors	
STATION	Sampling stations (S1 to S15)
BAND	Different distance to shore (B1=100 m, B2=100-250 m, B3=250-500 m, B4=500-1000 m)
YEAR	Consecutive years of sampling (year 1= annual cycle from october 2006 to october 2007; year 2= annual cycle from october 2007 to october 2008; year 3 and year 4 correspond to the wintering periods of 2009/2010 and 2010/2011 respectively)
Distance to disturbing or shelter elements	
DMMI	Distance to nearest MMCL island
DMEI	Distance to nearest Mediterranean island
DCAN	Distance to communication channel (Estacio Channel) with the Mediterranean Sea
DALB	Distance to main permanent discharge focus (Albujón wadi mouth)
DURB	Distance to nearest urban centre
DHARB	Distance to nearest harbour
DAIR	Distance to airport (San Javier Airport)
DCSAL	Distance to nearest saltpan (industrial area with shelter role for waterbirds)
DCWET	Distance to nearest natural wetland (protected area with shelter role for waterbirds)
Land uses (measured at 100 m, 1000 m and sub-basin scale)	
NDW	Natural dense wooded (*Acacia sp., Pinus sp.)*
NCW	Natural clear wooded (*Acacia sp., Pinus sp)*
NDS	Natural dense scrub (saline steppe and saltmarsh scrub)
NCS	Natural clear scrub (saline steppe and saltmarsh scrub)
DCW	Dry arboreal crops
DCS	Dry herbaceous crops
ICW	Irrigated arboreal crops (orchards)
ICS	Irrigated herbaceous and horticultural crops
UNP	Unproductive (urban areas)
WBS	Waterbodies (different pond types)

Table 1. Description of environmental variables

The first one comprised the areas of different land uses surrounding each sampling station (McKinney *et al*., 2006), at two spatial scales: 100 m buffer and 1000 m buffer discounting the previous 100 m buffer (1000 m ring hereafter), extracted from Martínez-López *et al.*, (2012). The second one included measures of distance to shelter/disturbing elements selected on the basis of previous work (Farinós and Robledano, 2010). All environmental variables were summarized using Principal Component Analysis (PCA) in order to detect redundancy between variables (Coreau and Martin, 2007) and to select the variables that best describe anthropogenic stress, landscape structure and functionality (Danz *et al*., 2007).

After selection and interpretation of nine PCs (table 2, figure in appendix1), environmental variables with a high correlation with an axis (Pearson´s coefficent > 0.6) were finally selected as representative descriptors of such gradient. Likewise, sampling stations were classified on the basis of their scores on such PCs through a MDS classification analysis (based on Euclidean distance) from a resemblance matrix. This resulted in four well defined groups (appendix 2), which were used as grouping factor in an Indicator Value Analysis (IndVal) based on a Monte Carlo test, in order to identify the most representative species of each group (on the basis of their relative abundance). This was performed with PC-Ord software (McCune and Mefford, 1999), separately for the winter and summer waterbird assemblages. Statistically significant indicator species (at $p < 0.05$) were selected as the candidate dependent variables to be included in multiple regression models (see following section).

Category	**PC**	**% variance**	**Gradient interpretation**	**Descriptor variables and correlation value with axis**
Land uses in 100 m buffer (PCA-b100)	1	33,2	Urban (-) vs Natural scrub (+)	WBS (-0.88), UNP (-0.87), NDS (0.69)
	2	23,46	Agro-rural (+) vs Dense wooded (-)	ICW (0.77), NDS (0.57)
	3	11,71	Short clear vegetation (+) vs High dense vegetation (-)	NCS (0.77), NDW (-0.65)
Land uses in 1000 m ring (PCA-r1000)	1	36,03	Agricultural (+) vs Heterogeneous wooded (-)	DCW (0.84), WBS (-0.75), DCS (0.72), NDW (-0.7), ICW (0.65), ICS (0.6)
	2	20,55	Urban (-) vs Natural scrub (+)	NDS (0.89), NCS (0.82), UNP (-0.5)
	3	15,13	Intensive humanization (-) vs Weak humanization (+)	NDW (0.58), WBS (0.57), NCW (0.56), UNP (-0.45)
Distance to disturbing or shelter elements (PCA-dist)	1	44,5	Funcional oceanic influence (-) vs Funcional continental influence (+)	DCWET (-0.88), DALB (-0.87), DCSAL (0.8), DAIR (-0.75), DMEI (0.7), DMMI (0.64)
	2	31,6	Structural oceanic influence (+) vs Structural continental influence (-)	DCAN (-0.82), DMMI (0.72), DMEI (-0.6), DHARB (0.54), DAIR (-0.53), DURB (0.51)
	3	13,4	Urban elements (+) vs Natural elements (-)	DURB (-0.65), DCAN (-0.53)

Table 2. Description of selected PCs and significant Pearson´s Correlation coefficents of input variables with each axis.

Multiple regression models

In order to analyze the relationship between calculated waterbird indices and environmental factors, multiple regression models were conducted for two different groups of dependent variables: (1) for ecological indices (TBU, R and H), the following steps were taken: first, the stations where indices' mean scores reached extreme values were identified; second, the axes with which these stations were associated were selected (from their scores in the PCA); third, the variables with a higher Pearson´s correlation coefficient with these axes (Pearson´s correlation coefficent > 0.6) were finally selected as input variables in GLMMs. (2) For indicator species' bird use, the groups of sampling stations (from Cluster classification of PCs and IndVal analysis) were the basis for selection: input variables were selected also when they showed a Pearson´s correlation coefficent > 0.6 with the axis (PC) to which sampling stations that shaped the group were associated. In relation to community indices, the ultimate criteria to perform a model were the results of the previous analyses of variance in relation to spatial factors: if significant differences were detected for an index, the regression analysis sought to identify the landscape or habitat factors contributing to these differences. Given the different distribution of the dependent variables (normal distribution in the case of Shannon Index, Poisson distribution in that of Total Bird Use, Richness and Bird Use of selected indicator species) two R packages were used: "nlme" (Pinheiro *et al.*, 2013) and "lme4" (Bates *et al.*, 2011), which include lme and glmer functions, used for normal distribution models (linear mixed models) and Poisson models (Generalized linear mixed models) respectively.

The procedure of model selection was based on lowering the Akaike information criterion (AIC, Burnham and Anderson, 2002) and including explanatory variables with statistically significant effect ($p < 0.05$). Three factors with spatial or temporal effects were included in the regression models, two as fixed grouping factors:

- BAND (to integrate the effect of distance to shore) and
- YEAR (to integrate the effect of time);

And the third one as a random grouping factor:

- STATION (to integrate the variability of inherent conditions of each sampling station not studied specifically in this paper).

Results

Spatio-temporal variation of waterbird-based indices

Waterbird census results are shown in appendix 3 as yearly means of bird use values for individual species and community indices. The statistical significance of temporal and spatial variation in waterbird indices is shown in table 3. Significant differences between winter and summer in three indices were consistent with the separate treatment of their respective waterbird communities.

	Annual	**Winter**				**Summer**			
Variables	**Season**	**Year**	**Sampling station**	**Band**	**Post-hoc "Band"**	**Year**	**Sampling station**	**Band**	**Post-hoc "Band"**
Total Bird Use[a]	<2.26e-16	6,30e-03	0.00078	4,30e-03	B1 *vs* B2 to B4	n.s.	0.04	1.88e-07	B1, B4 *vs* B2, B3
Richness	<2.26e-16	2,55e-02	0.0013	6,94e-08	B1 *vs* B2 to B4	0.0411	n.s.	6.39e-12	B1 *vs* B2 to B4
Shannon-Wiener Diversity	<2.26e-16	2,76e-02	0.012	0.00087	B1 *vs* B2 to B4	n.s.	n.s.	1.0e-13	B1 *vs* B2 to B4

Table 3. Significant results (p-values) of Kruskall-Wallis and Wilcoxon tests performed on waterbird community indexes in relation to temporal and spatial class variables. Wilcoxon Test was conducted for factors with only 2 classes: “season” (summer/winter) and “year” (only for the summer waterbird community, sampled twice). Kruskall-Wallis Test was conducted for factors with >2 classes: “station” and “band” in both seasons, and “year” (for the wintering waterbird community, sampled four times). n.s. = Non significant. The results of significant post-hoc comparisons are also shown

TBU, R and H showed significant differences for temporal (YEAR) and spatial factors (sampling station, BAND). In addition, linear regressions showed a decrease of three indices with BAND (lower values at higher distances from the shore): TBU (Adjusted R-squared=0.05, p=2.57e-05), R (Adjusted R-squared=0.25, p<2.26e-16) and H (Adjusted R-squared=0.148, p=1.17e-12.).

In summer, TBU showed significant differences between sampling stations as well as between bands; R showed significant differences between years and also between bands, whereas H only showed significant differences between bands. Linear regressions also showed a decrease of TBU (Adjusted R-squared=0.199, p=2.33e-06), R (Adjusted R-squared=0.507, p<2.2e-16), and H (Adjusted R-squared=0.55, p<2.2e-16) with BAND. Noteworthy are the results of post-hoc tests of differences in community structural indices (R and H) in relation to BAND, showing a significant change between B1 and the remaining bands (2-4). Also remarkably, in winter TBU does not differ significantly between B1 and B4, but with respect to B2 and B3, while in summer it behaves like other indices (B1 differs from the remaining bands). Finally, in both seasons, the MDS ordination confirmed that community composition changes markedly from B1 to B2-B4 (MDS stress=0.17 in both seasons). Thus, BAND stands as a key factor for community organization, which justifies to integrate it as a fixed grouping factor in subsequent multiple regression analyses.

Indicator species, community indices and influential gradients

The results of IndVal analyses for the four well-defined groups of sampling stations are shown in table 4.

Group	Stations	Description	Indicator species	IV significance	Season	Influential gradient
1	S15	Natural area exposed to oceanic (=Mediterranean) influence. Patches of *Pinus halepensis* and water bodies (saltpans); absence of structural disturbing elements of anthropogenic origin	*Larus genei*	0.021	Winter	PC2-b100, PC3-b100, PC1_r1000, PC2_r1000, PC2-dist, PC3-dist
2	S1, S2	Semi-natural area exposed to oceanic influence. Incipient urbanization; close to disturbing structural anthropogenic elements	*Sterna sandvicensis*	0.035	Winter	PC1-r1000, PC2-dist
3	S13, S14	Transitional area (urban-agricultural) of internal shore, influenced by land-based disturbing functional elements (point and diffuse effluent discharges); close to important disturbing structural anthropogenic elements (e.g. airport)	*Fulica atra*	0.036	Summer	PC1-r1000, PC1-dist
4	S10, S12	Inland shore with natural scrub (saline steppe, saltmarsh), influenced by land-based disturbing functional elements (point and diffuse effluent discharges); absence of disturbing structural anthropogenic elements.	*Podiceps cristatus* *Sterna albifrons* *Larus michaellis* *Eggretta garzetta*	0.003 0.005 0.034 0.008	Winter Summer Summer Summer	PC1-b100, PC3-b100, PC2-r1000, PC1-dist

Table 4. Sampling stations grouped by Cluster classification (structural-functional groups) from the first three axes of all the PCAs performed. Also shown are the Indicator Species for each group (from IndVal analyses), the season when the indicator value is statistically significant and the influential environmental gradients (PCs) for each species (clear association of preferred stations with axes).

Seven indicator species with statistical significance were identified, three in winter (*Larus genei*, *Sterna sandvicensis*, *Podiceps cristatus*) and four in summer (*Fulica atra, P. cristatus, Sterna albifrons, Larus michahellis, Egretta garzetta*), which were used as dependent variables in multiple regression models. Table 4 also shows the gradients *a priori* considered most influential for each group of stations on the basis of individual waterbird species' use. In the same way, table 5 shows the sampling stations where community indices reached extreme mean values, and the gradients (PCs) to which those stations were associated (from PCAs scores).

Index	Season	Sampling Stations	Value	Influential gradient
Bird Use	Winter	S12, S2	Max	PC1-b100, PC2-r1000
		S4	Min	PC1-dist, PC2-b100
	Summer	S12, S10,	Max	PC1-b100, PC2-r1000
		S4	Min	PC1-dist, PC2-b100
Richness	Winter	S12, S1	Max	PC2-b100
		S3, S4	Min	PC1-r1000, PC3-r1000, PC1-dist
Shannon Diversity	Winter	S11, S13	Max	PC1-dist
		S9, S3	Min	PC1-b100

Table 5. Waterbird community indices and the influential gradient affecting them as defined by the maximum and minimum values reached by these indices in sampling stations and by the clear association of these extreme stations with environmental gradients (from PCA interpretation). Gradients are only indicated when significant differences occur among stations for each index, otherwise models have not been performed.

Multiple regression models

The results of GLMMs are shown in table 6, which includes, for each model, the variables involved (in order of importance) and the total deviance explained (%). Following our final selection criteria, multiple regression models for R and H in summer were not considered since there were no significant differences for any of these indices between sampling stations.

MODEL	Explained deviance	Factor classes observations
Winter		
Total Bird Use~Band + Year	20.41 %	(-)B2 (-)B3 (-)B4 (-)Year2 (-)Year3 (-)Year4
Richness~Band + Year + Nds_b100 - Nds_b100^2 - Icw_b100	39.92 %	(-)B2 (-)B3 (-)B4 (+)Year2 (+)Year3 (+)Year4
Shannon Diversity~Band + Year	9.16 %	(-)B2 (-)B3 (-)B4 (+)Year2 (+)Year3 (+)Year4
Use of *Podiceps cristatus*~Band + Year + Nds_r1000 - Dmmi+ Dmmi2	57.42 %	(+)B2 (+)B3 (+)B4 (-)Year2 (+)Year3 (+)Year4
Use of *Larus genei*~Band + Year + Ndw_r1000 + Ncs_b100 – Ncs_b1002	81.8 %	(-)B2 (-)B3 (-)B4 (-)Year2 (-)Year3 (+)Year4
Use of *Sterna sandvicensis*~Band + Year - Dcan	44.39 %	(-)B2 (-)B3 (-)B4 (+)Year2 (+)Year3 (-)Year4
Summer		
Total Bird Use~Band + Year + Dcsal + Wbs_r1000 - Wbs_r1000^2	51.25 %	(-)B2 (-)B3 (-)B4 (+)Year2
Use of *Fulica atra*~Band - Year + Nds_r1000 - Ics_r1000 + Ics_r10002	44.16 %	(-)B2 (-)B3 (-)B4 (+)Year2
Use of *Sterna albifrons*~Band + Year - Dcwet + Dmmi + Ncs_b100	89.53 %	(-)B2 (-)B3 (-)B4 (-)Year2
Use of *Larus michaellis*~Band + Year + Ncs_b100	36.98 %	(-)B2 (-)B3 (-)B4 (+)Year2
Use of *Eggretta garzetta*~Band + Year + Nds_b100 + Ncs_r1000	57.76 %	(-)B2 (-)B3 (-)B4 (-)Year2

Table 6. Multiple regresion models of indices and indicator species

Complementarily, appendix 4 shows the value of the β coefficient and the sign affecting each dependent variable (each factor class in the case of categorical variables). It is inevitable to highlight the weight in all models of spatial and temporal variation, illustrated by BAND and YEAR, the first two terms integrated (in that order) through the stepwise procedure, and those accounting for the largest proportion of explained deviance.

Discussion

Spatial and temporal variation of waterbird-based indices

Significant differences between years, sampling stations and bands appear in all cases in winter indices. YEAR expresses temporal changes in waterbird populations reflected in local surveys, but partly attributable to processes operating at larger scales (e. g. reproductive success, migration patterns, survival; Tománková *et al.*, 2013), and partly to local inter-annual changes (Lopes *et al*., 2005; Rendón *et al*., 2008). Since BAND is a measure of the distance to attractive or deterrent shoreline features, the decrease of H and R from B1 to B2-B4 suggests that winter community structure is affected by them, as well as by depth (inversely related with BAND), which determines the distribution of morphological types, feeding styles and foraging strategies (Liordos, 2010; Athearn *et al*., 2012). In addition, several species exploiting the first band (herons, terns, coot, etc.) are attracted by specific food patches located in shallow areas, i.e. macrophyte meadows, algal mats and their associated invertebrate and fish communities (Verdiell-Cubedo *et al*., 2008; 2012). In summer, significant differences were found only between bands for all indices, and in one case between stations (TBU) or years (R). The general decline in the value of indexes with respect to winter and the remarkable absence of differences in R and H between stations are indicative of community impoverishment and homogenization. Variation of TBU is probably due to increased stress from seasonal tourism in urban-affected areas (Farinós and Robledano, 2010), which force summer species to concentrate in less anthropogenic ones.

The overwhelming influence of BAND and YEAR does not imply that the responses to other variables should be disregarded, but that their effects must be isolated from local environmental factors. In any case, a great amount of unexplained variance is a frequent result when modeling waterbird responses to wetland characteristics (Pescador *et al.*, 2012).

Community indices' models

The negative response to BAND of both winter and summer TBU and R is consistent with the eco-morphological requirements of species. Band 1 offers a greater variety of foraging niches and food resources on which different strategies and morphotypes can coexist, but that is reduced with distance as depth increases. Higher β in the response of summer TBU indicates a lower abundance of waterbirds in deeper waters with respect to winter, probably related to the increase of water-based recreational activities (Farinós and Robledano, 2010). Maximum winter values of TBU and R were recorded in stations of transitional environments with very pronounced gradients: continental-lagoon in the main wadi mouth (S12), and lagoon-marine in in areas adjacent to sea openings (S2 and S1). Although they are structurally and functionally very different, all three areas probably offer a high diversity of trophic resources, rich food patches and more feeding niches. Furthermore, areas where TBU peaks in summer (S12, S10) are far from saltpans and their disturbed vicinity, hence waterbirds would prefer these natural shorelines with less human presence (positive response of summer TBU to DCSAL). Moreover, they are close to alternative inland feeding habitats (natural and restored ponds, old sewage works, irrigation ponds; Sebastián-González *et al.*, 2010), which is reflected in a positive quadratic response of summer TBU to the presence of waterbodies in the first 1000 m inland (WBS_ r1000).

There is a positive quadratic response of winter R to well preserved salt steppe or saltmarsh (NDS) in the first 100 m of land, which is the original, structurally undisturbed landscape of the MMCL inland shoreline (Carreño *et al*., 2008). This agrees with the results of Palacio-Núñez *et al*. (2007), where less human-impacted and structurally more heterogeneous stations had higher bird richness and diversity. By contrast, the negative response to irrigated arboreal crops in the 100 m buffer (ICW_b100) suggests a negative functional influence (subsurface diffusion of agricultural drainage causing eutrophication), or a negative perception of vegetation structure (disruption of the traditional open landscape). Both effects are compatible with a reduction of bird richness through the displacement of the less tolerant species.

The gradient of increasing diversity related with proximity to the Mediterranean sea, recognized in the MMCL for several other aquatic taxa (Rosique, 2000) has been proposed by Farinós and Robledano (2010) as an explanation for a similar gradient in the diversity of the offshore waterbird community. This richness/diversity gradient is also recognized in the present study (high R in S2 and max in S1), and by integrating the influence of terrestrial habitats, spatially extended (richness and/or diversity are also favored by natural landscapes, e.g. NDS) (Tables 5 and 6). The most diverse areas for waterbirds share the characteristics of being more natural and closer to the open sea. Only S2 is a low quality habitat based both on benthic indicators (Salas *et al*., 2006) and on terrestrial features (heavily urbanized), but its openness to the main sea could offset these limitations, leading to high waterbird diversity. Therefore, the disconnection from the Mediterranean would explain the minimum waterbird value of other sections with similar impacts (S3 and S4).

Wintering indicator species' models

On the basis of different ecomorphological requirements, waterbirds respond specifically to distance to shore: positively in the case of *P. cristatus* (diver) and negatively in that of *S. sandivcensis* and *L. genei* (shallow water feeders). *P. cristatus* also responds positively to NDS in the first 1000 m. Verdiell-Cubedo *et al.*, (2012) reported high densities and biomasses of littoral (in S12) and bentic fishes (in S10 to S12), which could benefit piscivorous species like *P. cristatus*. However, its preference for natural stations (S10, S12) subject to the diffusion of agricultural drainage, could manifest a background response to the functional influence of landscape processes (i.e. agricultural pollution) previously suggested by Robledano *et al.* (2011). On the other hand, *L. genei* responds positively to NCS in the first 100 m and to NDW in the subsequent 900 m (1000 m ring). In practice this represents sparse saline steppe and saltmarsh habitats with a second vegetation belt composed of tall helophytes - favored by agricultural irrigation -, invasive *Acacia sp.* or *Pinus halepensis* woodlots (like in S15, where *L. genei* is indicator species). Without ignoring the conservation implications of invasive trees, this habitat combination should be regarded as the species' landscape template. Finally, *S. sandvicensis* is tied to areas closest to the functional lagoon mouth: S1 (natural section adjacent to S15, a protected area), and S2 (more disturbed urban section). It seems more influenced by the trophic advantage represented by the proximity and communication with open waters (Farinós and Robledano, 2010), belonging to an adjacent Important Bird Area (IBA 407) reported as a valuable feeding area for seabirds (Infante *et al.,* 2011).

Summer indicator species' models

The negative response of all indicator species to distance to shore is due to their preference for shallow feeding areas rich in food resources for both phytophagous and fish-eating species. The indicator character of *F. atra* in summer in S13 and S14 seems inconsistent with its phenology and habitat preferences, but can be explained by the concentration of part of the population in traditional palustrine habitats of the inner shore during transition months (April, October). *F. atra* also presents a positive quadratic response to irrigated herbaceous crops in the first 900 m ring of land, for which it has been attributed and indicator role of eutrophication (Robledano *et al.*, 2008, 2011). *S. albifrons* shows a strong negative response to the distance to shore (reflected by most of the model's deviance explained by BAND), regardless of shore habitat naturalness. For other tern species Farinós and Robledano (2010) pointed to a low specific sensitivity to human disturbance and to aerial foraging (Blumstein *et al.*, 2003), as possible explanations for their distribution. The marginal effects of other variables deserve some additional comments, highlighting the negative response to distance to the nearest wetland and the positive response to NCS in the first 100 m. As indicator species of S10 and S12 (natural stations near wetlands like marshes with ponds), its reluctance to move away from the shoreline can also reflect some preference for such habitats.

L. michahellis shows a more littoral distribution, less conditioned by the presence of elements like islands and favoring natural stations (Table 6). Following its generalist and opportunistic feeding behaviour, this species favors areas with higher waterbird species richness, probably a clue to a greater diversity of foraging opportunities (including egg and chick predation or kleptoparasitism). Finally, *E. garzetta*, as a wading species, shows a negative response to distance to shore and a positive one to natural scrub habitats, either well-preserved (NDS) or slighty disturbed (NCS). The difference in vegetation cover between these two habitat types, however, does not necessarily indicate conservation status, but natural ecological character or successional stage. Thus, the landscape preferences of Little Egret include low plant cover habitats (saline steppe and open saltmarsh), although more concealed immediate shoreline areas seem also essential (dense scrub).

Management implications

On the basis of our results, we can raise some management and research guidelines to enhance waterbird diversity and conservation value in coastal lagoons: (1) to give effective protection and promote the restoration of well-preserved natural habitats, emphasizing the maintenance of their structural continuity inland and taking advantage of the functional effects of the open sea-lagoon mutual influences; (2) to reduce disturbance in the first band of nearshore waters, where due to different ecomorphological and behavioural strategies, several relevant species converge; (3) to combine the preservation and enhancement of the natural physiognomy of the shoreline, with the management of certain land-based impacts whose functional effects reach waterbirds via 'hidden pathways' (e.g. subsurface hydrological processes, lagoon trophic webs). An example is eutrophication caused by excessive fertilization altering waterbird assemblages in the long-term (Robledano *et al.*, 2011). In this respect, it would be interesting to implement and test on the field the landscape-scale scheme proposed in Farinós *et al.* (2013), for dealing with agricultural effluents flowing into the lagoon's basin from its watershed. Essentially, it proposes a double set of man-made wetlands: an inland one with treatment functions, which would improve water quality (mitigating background eutrophication), and a littoral one, which would recover structurally valuable shoreline habitats. And (4), to take into account previous research on the role of natural habitat windows (saltmarshes, saline steppes, reedbeds), of proven value for nearshore waterbirds, for other relevant avian assemblages like steppe birds (Robledano *et al.*, 2010), searching for positive conservation synergies.

Concluding remarks

In semi-enclosed coastal systems, the richness and abundance of waterbirds can be favored under moderate degradation states (i.e. eutrophication), but usually within well-preserved natural environments (NDS, NCS). The influence of such natural features, as well as that of internal lagoon gradients (e.g. openness, confinement) overrides the secondary effects of anthropogenic landscape impacts. High waterbird richness and the presence of indicator species of conservation concern (i.e. *L. genei*, *S. sandvicensis*; Birdlife International, 2004) signal the structural and functional features of the best preserved lagoon habitats, including healthy adjacent landscapes providing refuge or buffering land-based impacts. Therefore, the control of eutrophication is expected to favor specialist waterbird species (Bryce *et al*., 2002), usually of great conservation value, while reducing the abundance of generalist ones. In any case, general conservation value is expected to rise. The consideration of both types of responses (general to natural areas, and local to disturbing processes) enhances the value of waterbirds as integrative environmental signals compared to other bioindicators (e.g. benthic biocoenoses, fish assemblages), which nevertheless should complement the assessment of local ecological condition. The difficulty of implementing other indices (i.e. those based on benthos) in Mediterranean CLs, is the high variability in the composition of species (Pérez-Ruzafa *et al*., 2011; García-Sánchez *et al*., 2012), making it difficult to extrapolate the detected patterns and associations. Waterbird assemblages of Mediterranean CLs are more homogeneous and consistent among sites across relatively large geographical divisions (see http://wpe.wetlands.org/). Assuming that the same major environmental drivers, both natural (hydrological, geomorphological) and anthropogenic (agriculture, fishery management, urbanization), govern the functioning of Mediterranean CLs, we can expect waterbird species and communities to respond in a similar way. Long-term datasets provided by international waterbird censuses (IWC) in representative CLs will be amenable to comparative studies of sites subjected to varying degrees of disturbance (i.e, in different states of alteration with respect to a ‘good’ reference state). Given the potential utility of such an approach, it is surprising the superficiality with which the ornithological value of coastal lagoons is presented in global assessments (Newton *et al*., 2014). In fact, the usual bottom-up approach to the MMCL ecology and management (Lloret and Marin, 2009; Marín-Guirao *et al*., 2008, Pérez-Ruzafa *et al*., 2002; 2005) often neglects the waterbird component. So it becomes necessary to use waterbirds as bioindicators in semi-enclosed coastal systems like MMCL by integrating the analysis of their role as ecological indicators of lagoon status (analysing their variations in relation to food web ascending effects) and their dependence on the nature of adjacent terrestrial habitats. Such eco-hydromorphological approach is in fact not new to the WFD’s assessment of rivers, including adjacent riparian systems (Raven *et al*., 2010), or transitional waters (Borja, 2005).

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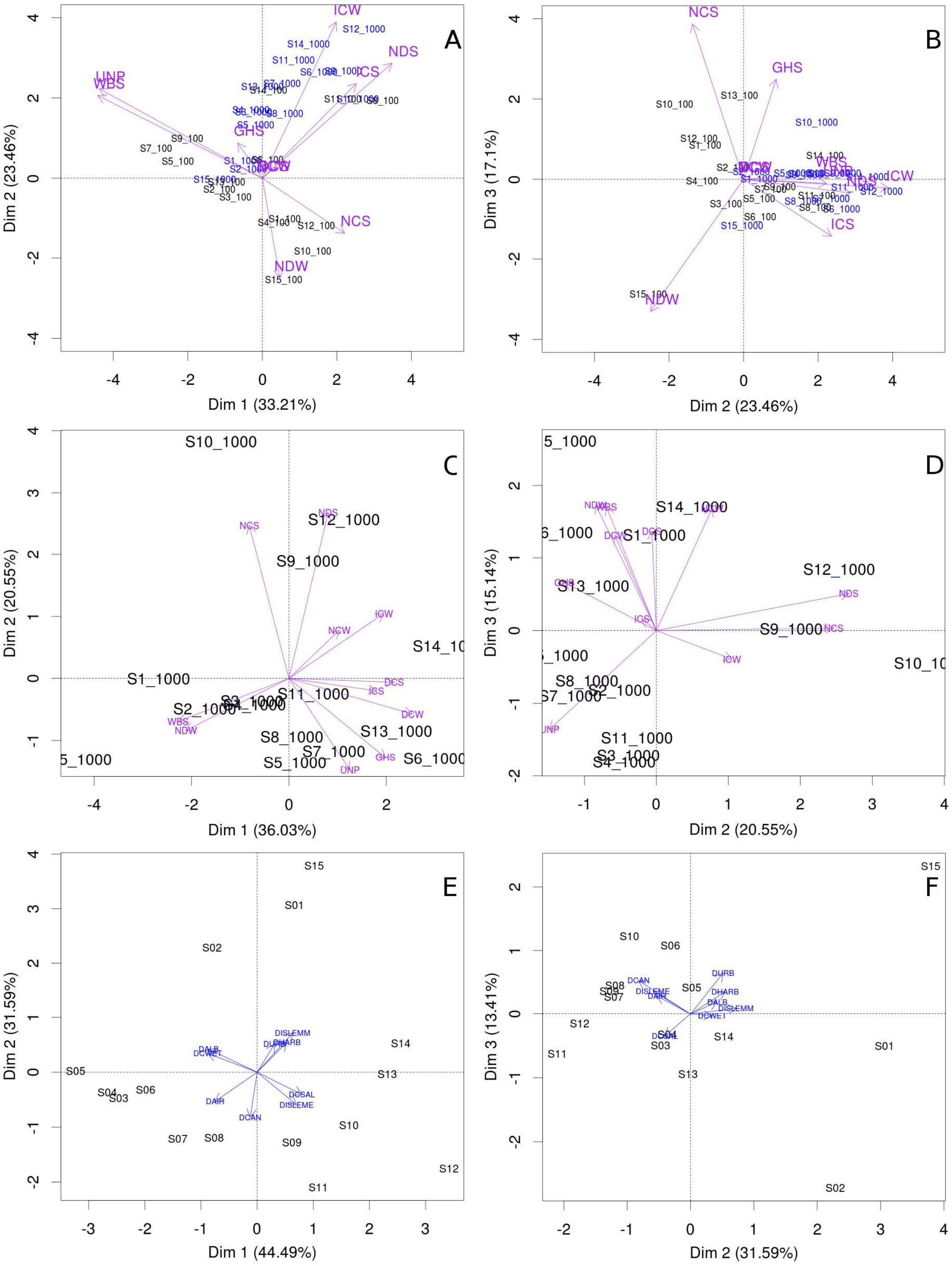
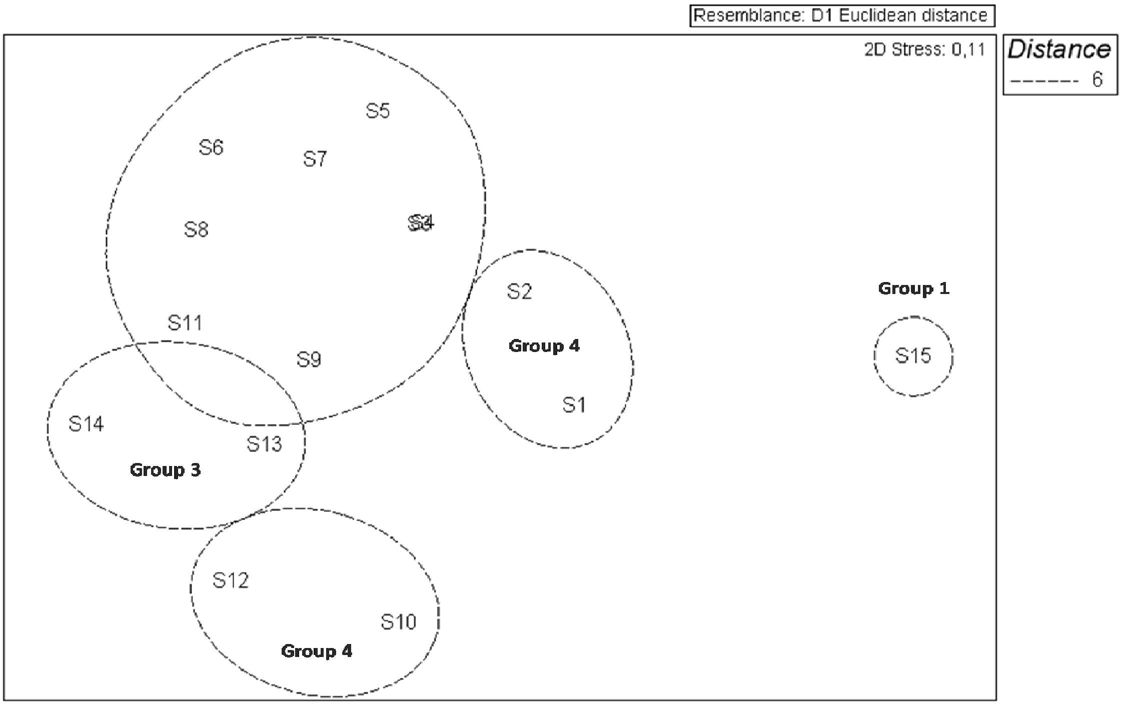
Appendices

Appendix 1. Principal Component Analysis bi-plot representing different scales and gradients: surfaces of each land use type (in ha) measured in 100 m buffer and showing land uses surfaces in 1000 m as supplementary cases (graph A: PC1 and PC2; B: PC2 and PC3); surfaces of each land use type (in ha) measured in 1000 m ring (graph C: PC1 and PC2; D: PC2 and PC3); and distance to shelter or stressful elements (graph E: PC1 and PC2; F: PC2 and PC3).



Appendix 2. Classification of sampling stations from PCs scores (three first axes of each analysis: 100 m buffer, 100 m ring and distance) on the basis of Euclidean distances from the resemblance matrix.



Appendix 3. Waterbird census in study area from 2006 to 2011.Winter values in each case (band*station) have been averaged from 2006 to 2008 and 2009 to 2011 winter season records and summer values have been averaged from 2006 to 2008 summer season records. Species data is expresed as bird use (birds*day-1)

	FUAT	POCR	PONI	PHCA	MESE	TATA	ANPL	STSA	STHI	STAL	LACA	LAGE	LAAU	LARI	EGGA	EGAL	ARCI	PHRO	TBU	Richness	Shannon Diversity
Winter S1B1	0	22,5	15	930	0	0	127,5	367,5	150	0	1335	457,5	15	465	210	0	37,5	450	4582,5	6,75	1,627005143
Winter S1B2	0	90	60	30	0	0	0	120	0	0	697,5	22,5	0	0	0	0	0	0	1020	4,25	0,796899825
Winter S1B3	0	37,5	292,5	60	0	0	0	22,5	0	0	412,5	0	0	15	0	0	0	0	840	3,25	0,730465118
Winter S1B4	0	67,5	367,5	82,5	0	0	0	0	0	0	255	0	0	0	0	0	0	0	772,5	3,25	0,757462469
Winter S2B1	0	15	60	142,5	0	0	0	90	0	0	2072,5	427,5	495	1440	30	0	0	0	5572,5	6	0,965046471
Winter S2B2	0	15	390	52,5	0	0	0	97,5	0	0	667,5	52,5	0	547,5	0	0	0	0	1822,5	4,5	1,193711303
Winter S2B3	0	67,5	637,5	52,5	0	0	0	0	0	0	615	7,5	0	915	0	0	0	0	2295	4,5	0,910359264
Winter S2B4	0	7,5	315	15	0	0	0	0	0	0	97,5	7,5	0	0	0	0	0	0	442,5	2,5	0,534355779
Winter S3B1	0	45	52,5	90	30	0	0	15	0	0	405	7,5	0	225	7,5	0	0	0	877,5	4,75	1,146649279
Winter S3B2	0	30	262,5	105	202,5	0	0	0	0	0	135	0	0	7,5	0	0	0	0	742,5	3	0,807049505
Winter S3B3	0	97,5	390	675	22,5	0	0	0	0	0	2430	0	0	0	0	0	0	0	3615	4	1,001774172
Winter S3B4	0	30	225	615	45	0	0	0	0	0	2032,5	0	0	0	0	0	0	0	2947,5	4	0,882828223
Winter S4B1	0	0	0	15	0	0	0	0	0	0	915	30	0	67,5	22,5	0	7,5	0	1057,5	3,75	0,709736737
Winter S4B2	0	0	7,5	15	0	0	0	7,5	0	0	322,5	0	0	0	7,5	0	0	0	360	2,25	0,532827654
Winter S4B3	0	37,5	7,5	7,5	22,5	0	0	0	0	0	135	0	0	0	0	0	0	0	210	1,5	0,452882262
Winter S4B4	0	37,5	225	52,5	0	0	0	0	0	0	7,5	0	0	0	0	0	0	0	322,5	2	0.738617756
Winter S5B1	0	15	90	60	7,5	0	0	0	0	0	345	75	0	322,5	7,5	0	0	0	922,5	5,25	1,185394404
Winter S5B2	0	90	157,5	7,5	45	0	0	0	0	0	97,5	0	0	37,5	0	0	0	0	435	3,25	1,021305093
Winter S5B3	0	142,5	337,5	30	0	0	0	0	0	0	142,5	0	0	30	0	0	0	0	682,5	3,5	0,964274534
Winter S5B4	0	450	900	52,5	15	0	0	0	0	0	487,5	7,5	0	7,5	7,5	0	0	0	1927,5	4,75	1,213037609
Winter S6B1	0	37,5	82,5	172,5	30	0	0	15	0	0	142,5	82,5	0	7,5	45	7,5	0	0	622,5	5	1,122771769
Winter S6B2	0	150	292,5	15	0	0	0	7,5	0	0	52,5	0	0	52,5	0	0	0	0	570	3,75	1,124062961
Winter S6B3	0	172,5	540	0	0	0	0	0	0	0	30	0	0	37,5	0	0	0	0	780	2,75	0,786468848
Winter S6B4	0	607,5	1387,5	30	105	0	0	0	0	0	195	0	0	7,5	0	0	0	0	2332,5	4,25	0.960902848
Winter S7B1	0	7,5	67,5	0	0	0	0	0	0	0	52,5	75	0	172,5	7,5	0	0	0	382,5	3,25	0,949251583
Winter S7B2	0	52,5	60	0	0	0	0	0	0	0	22,5	0	0	0	0	0	0	0	135	2,25	1,009542311

	FUAT	***POCR***	***PONI***	***PHCA***	***MESE***	***TATA***	***ANPL***	***STSA***	***STHI***	***STAL***	***LACA***	***LAGE***	***LAAU***	***LARI***	***EGGA***	***EGAL***	***ARCI***	***PHRO***	***TBU***	***Richness***	***Shannon Diversity***
Winter S7B3	0	217,5	307,5	15	0	0	0	0	0	0	187,5	0	0	0	0	0	0	0	727,5	3	1,017163327
Winter S7B4	0	1012,5	892,5	82,5	0	15	0	0	0	0	292,5	0	0	7,5	7,5	0	0	0	2310	4,5	1,140048943
Winter S8B1	0	45	135	15	0	0	0	0	0	7,5	90	90	0	37,5	45	0	7,5	0	472,5	5,75	1,624920775
Winter S8B2	0	165	180	0	0	0	0	0	0	30	52,5	0	0	120	0	0	0	0	547,5	3,5	1,104090305
Winter S8B3	0	262,5	375	0	0	0	0	0	0	22,5	67,5	0	0	30	0	0	0	0	757,5	3,25	1,03835419
Winter S8B4	0	1065	1132,5	67,5	105	0	0	0	0	0	105	0	0	0	7,5	0	0	0	2482,5	4,75	1,123611591
Winter S9B1	0	0	0	3262,5	0	0	0	0	0	0	105	52,5	0	0	52,5	0	0	0	3472,5	2,25	0,359033236
Winter S9B2	0	37,5	195	7,5	0	0	0	0	0	0	135	0	0	0	0	0	0	0	375	2,5	0,598090882
Winter S9B3	0	405	300	30	0	0	0	0	0	0	262,5	0	0	0	0	0	0	0	997,5	3,25	1,038328173
Winter S9B4	0	1927,5	735	30	0	0	0	0	0	0	540	0	0	0	7,5	0	0	0	3240	3,75	0,951894717
Winter S10B1	0	15	15	67,5	0	22,5	0	97,5	0	0	540	105	0	135	45	22,5	0	0	1065	5,75	1,309923572
Winter S10B2	0	82,5	202,5	2347,5	52,5	0	0	0	0	0	225	0	0	0	0	0	0	0	2910	3,5	0,790836313
Winter S10B3	0	517,5	405	52,5	0	0	0	0	0	0	172,5	0	0	0	0	0	0	0	1147,5	3,25	1,028693368
Winter S10B4	0	1762,5	720	112,5	0	0	0	7,5	7,5	0	487,5	7,5	0	22,5	15	0	0	0	3142,5	4,75	0,992023787
Winter S11B1	15	30	45	135	0	67,5	0	127,5	0	0	472,5	330	0	675	127,5	15	7,5	52,5	2100	8,5	1,740671107
Winter S11B2	0	127,5	157,5	7,5	0	0	0	22,5	7,5	0	315	7,5	0	30	0	0	0	0	675	3,5	0,881337866
Winter S11B3	0	172,5	172,5	60	0	0	0	0	0	0	82,5	0	0	0	0	0	0	0	487,5	1,5	0,993064117
Winter S11B4	0	810	142,5	15	0	0	0	0	0	0	22,5	7,5	0	0	7,5	0	0	0	1005	2,25	0,428224218
Winter S12B1	5377,5	60	75	15	0	15	22,5	0	15	0	330	225	0	555	97,5	15	60	0	6862,5	8,25	1,066898264
Winter S12B2	0	502,5	352,5	7,5	0	0	0	0	0	0	210	7,5	0	682,5	0	0	0	0	1762,5	4	0,852345419
Winter S12B3	0	1695	577,5	22,5	0	0	60	30	0	0	135	0	0	0	0	0	0	0	2520	4	0,964250042
Winter S12B4	0	915	1155	390	0	0	45	15	0	0	232,5	0	0	0	0	0	0	0	2752,5	4,5	1,185847818
Winter S13B1	1267,5	75	7,5	240	0	0	0	307,5	0	0	412,5	60	7,5	450	37,5	0	0	0	2865	6,25	1,062842972
Winter S13B2	0	120	172,5	67,5	0	0	0	37,5	0	0	67,5	0	0	0	0	0	0	0	465	3,5	1,029228587
Winter S13B3	15	172,5	360	7,5	15	0	0	7,5	0	0	22,5	0	0	0	0	0	0	0	600	3,25	0,768089454
Winter S13B4	0	495	1185	127,5	0	0	0	0	0	0	82,5	0	0	0	0	0	0	0	1890	3,25	0,82625209
Winter S14B1	1012,5	60	22,5	30	0	0	0	0	0	0	345	45	7,5	7,5	30	0	15	0	1575	5,75	1,045017222

	FUAT	**POCR**	**PONI**	**PHCA**	**MESE**	**TATA**	**ANPL**	**STSA**	**STHI**	**STAL**	**LACA**	**LAGE**	**LAAU**	**LARI**	**EGGA**	**EGAL**	**ARCI**	**PHRO**	**TBU**	**Richness**	**Shannon Diversity**
Winter S14B2	0	75	22,5	75	0	0	0	7,5	7,5	0	120	0	0	0	0	0	0	0	307,5	3	0,811818214
Winter S14B3	0	75	172,5	15	0	0	0	0	0	0	45	0	0	0	0	0	0	0	307,5	2,25	0,908644887
Winter S14B4	0	322,5	517,5	15	0	0	0	0	0	0	510	0	0	0	0	0	0	0	1365	3,5	1,001118623
Winter S15B1	0	0	0	322,5	0	0	0	0	0	0	1080	667,5	0	675	15	82,5	0	0	2842,5	4,5	1,273359031
Winter S15B2	0	0	7,5	15	15	0	37,5	0	0	0	187,5	15	0	157,5	0	0	0	187,5	622,5	3	0,564264327
Winter S15B3	0	37,5	0	7,5	52,5	0	0	15	0	0	120	0	0	0	0	0	0	0	232,5	1,75	0,643623323
Winter S15B4	0	15	22,5	112,5	90	0	0	0	0	0	285	0	0	0	0	0	0	0	525	3	0,850240655
Summer S1B1	0	0	0	0	0	0	0	15	495	270	450	120	60	45	180	0	15	60	1710	7	1,636612536
Summer S1B2	0	0	0	0	0	0	0	0	15	45	75	480	0	0	0	0	0	0	615	2,5	0,651956367
Summer S1B3	0	0	0	0	0	0	0	0	0	0	135	0	0	105	0	0	0	0	240	1	0
Summer S1B4	0	0	15	0	0	0	0	0	0	0	60	0	0	0	0	0	0	0	75	1	0
Summer S2B1	0	0	15	0	0	0	0	60	150	45	165	15	0	75	0	0	0	0	525	5	1,392684438
Summer S2B2	0	0	180	0	0	0	0	0	45	0	195	0	0	0	0	0	0	0	420	2,5	0,795926992
Summer S2B3	0	0	300	0	0	0	0	0	0	0	135	0	0	0	0	0	0	0	435	2	0,430116122
Summer S2B4	0	0	90	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	90	1	0
Summer S3B1	0	0	30	0	0	0	0	30	15	0	165	0	0	30	0	0	0	0	270	3,5	0,902138976
Summer S3B2	0	0	15	0	0	0	0	0	0	0	345	0	0	0	0	0	0	0	360	1,5	0,089422456
Summer S3B3	0	0	0	0	0	0	0	0	0	0	105	0	0	0	15	0	0	0	120	1	0,188385081
Summer S3B4	0	15	0	0	0	0	0	0	0	0	780	0	0	0	0	0	0	0	795	1,5	0,318257084
Summer S4B1	0	0	0	0	0	0	0	0	0	0	45	30	0	30	0	0	0	0	105	2	0,599424656
Summer S4B2	0	0	0	0	0	0	0	0	0	0	15	0	0	0	0	0	0	0	15	0,5	0
Summer S4B3	0	0	30	0	0	0	0	0	0	0	15	0	0	0	0	0	0	0	45	1	0,318257084
Summer S4B4	0	0	0	0	0	0	0	0	0	0	15	0	0	0	0	0	0	0	15	0,5	0
Summer S5B1	0	0	0	0	0	0	0	0	0	0	30	0	0	135	30	0	0	0	195	2,5	0,694276433
Summer S5B2	0	15	15	0	0	0	0	0	15	0	15	0	0	15	0	0	0	0	75	2,5	0,693147181
Summer S5B3	0	15	75	0	0	0	0	0	0	0	15	0	0	0	0	0	0	0	105	2	0,47513527
Summer S5B4	0	0	0	0	0	0	0	0	0	0	15	0	0	0	0	0	0	0	15	0,5	0
Summer S6B1	0	0	0	0	0	0	0	60	15	60	90	0	0	0	0	0	0	0	225	3	0,954771252
Summer S6B2	0	0	0	0	0	0	0	30	0	0	0	0	0	0	0	0	0	0	30	0,5	0
Summer S6B3	0	0	0	0	0	0	0	0	0	0	45	0	0	0	0	0	0	0	45	0,5	0

	FUAT	POCR	PONI	PHCA	MESE	TATA	AMPL	STSA	STHI	STAL	LACA	LAGE	LAAU	LARI	EGGA	EGAL	ARCI	PHRO	TBU	Richness	Shannon Diversity
Summer S6B4	0	0	0	0	0	0	0	0	0	0	30	0	0	0	0	0	0	0	30	0,5	0
Summer S7B1	0	0	0	0	0	0	0	0	0	15	30	0	0	45	15	0	0	0	105	2,5	0,82170886
Summer S7B2	0	0	0	0	0	0	0	0	0	0	15	0	0	0	0	0	0	0	15	0,5	0
Summer S7B3	0	0	0	0	0	0	0	0	15	0	15	0	0	0	0	0	0	0	30	1	0
Summer S7B4	0	0	0	0	0	0	0	0	0	0	30	0	0	0	15	0	0	0	45	1,5	0,34657359
Summer S8B1	0	0	0	0	0	0	0	15	15	90	0	30	0	45	45	0	0	0	240	4	1,21081561
Summer S8B2	0	0	0	0	0	0	0	0	0	0	30	0	0	15	0	0	0	0	45	1	0
Summer S8B3	0	15	0	0	0	0	0	0	0	0	60	0	0	0	0	0	0	0	75	1,5	0,318257084
Summer S8B4	0	0	0	0	0	0	0	0	0	0	15	0	0	0	60	0	0	0	75	1	0
Summer S9B1	0	0	0	0	0	0	0	0	30	30	45	0	0	15	0	0	0	0	120	2,5	0,045717160
Summer S9B2	0	0	480	0	0	0	0	0	0	0	45	0	0	0	0	0	0	0	525	1	0,146254234
Summer S9B3	0	150	15	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	165	1	0,152318049
Summer S9B4	0	150	15	0	0	0	0	0	0	0	0	0	0	0	75	0	0	0	240	1,5	0,318257084
Summer S10B1	0	0	0	0	0	0	0	405	105	1080	1680	60	0	15	165	0	0	0	3510	6,5	1,294213209
Summer S10B2	0	0	75	0	0	0	0	0	0	0	30	0	0	0	15	0	0	0	120	1,5	0,318257084
Summer S10B3	0	15	0	0	0	0	0	0	0	0	15	0	0	0	0	0	0	0	30	1	0
Summer S10B4	0	0	0	0	0	0	0	0	0	0	30	0	0	0	15	0	0	0	45	1	0,318257084
Summer S11B1	0	0	0	0	0	0	0	15	285	180	315	45	0	15	135	0	15	0	1005	6,5	1,517183252
Summer S11B2	0	0	0	0	0	0	0	0	0	0	60	0	0	0	0	0	0	0	60	1	0
Summer S11B3	0	0	0	0	0	0	0	0	0	0	90	0	0	0	0	0	0	0	90	0,5	0
Summer S11B4	0	0	75	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	75	0,5	0
Summer S12B1	30	45	0	0	0	0	0	45	75	870	1365	0	0	135	195	0	45	0	2805	7	1,206086865
Summer S12B2	0	90	0	0	0	0	0	0	15	0	165	0	0	30	0	0	0	0	300	3	0,903755647
Summer S12B3	0	15	0	0	0	0	0	0	30	15	210	0	0	210	0	0	0	0	480	3	0,560905883
Summer S12B4	60	0	0	0	0	0	0	0	30	0	60	0	0	0	0	0	0	0	150	1,5	0,527460004
Summer S13B1	0	0	0	0	0	0	0	0	0	90	345	15	0	180	15	0	0	0	645	3,5	0,854742955
Summer S13B2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Summer S13B3	30	0	0	0	0	0	0	0	0	0	45	0	0	0	0	0	0	0	75	1	0,336505834
Summer S13B4	0	0	0	0	0	0	0	15	0	0	165	0	0	0	0	0	0	0	180	1	0,143417992
Summer S14B1	165	0	0	0	0	0	0	0	0	120	255	15	0	15	0	0	0	0	570	4	1,175377709
Summer S14B2	0	0	0	0	0	0	0	0	0	0	45	0	0	0	0	0	0	0	45	0,5	0
Summer S14B3	0	0	0	0	0	0	0	0	0	0	30	0	0	0	0	0	0	0	30	0,5	0
Summer S14B4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Summer S15B1	0	0	0	0	0	0	0	0	90	45	210	0	0	90	0	0	0	0	435	3,5	1,098181941
Summer S15B2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Summer S15B3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Summer S15B4	0	0	0	0	0	0	0	0	0	0	150	0	0	0	0	0	0	0	150	1	0

Appendix 4. Detailed output of GLMMs for each index and species for winter and summer season. Estimated β coefficients for explanatory variables are shown if the p-value was minor than 0.05. Each model was selected under the criteria of the lowest AIC.

	Models for winter season						Models for summer season				
Variable	Total bird use	Richness	Shannon Index	*Podiceps cristatus*	*Larus genei*	*Sterna sandvicensis*	Total bird use	*Fulica atra*	*Sterna albifrons*	*Larus michaellis*	*Egretta garzetta*
Band 2	-1.005	-0.506	-0.277	1.296	-3.272	-1.223	-1.558	-0.1954	-4.164	-1.612	-3.951
Band 3	-0.724	-0.542	-0.262	2.301	-5.788	-2.568	-1.847	-1.872	-5.263	-1.736	-3.951
Band 4	-0.244	-0.404	-0.239	3.107	-4.524	-3.811	-1.840	-1.179	-21.03	-1.346	-1.553
Year 2	-0.595	0.046	0.149	-0.845	-0.99	0.481	0.303	1.03	-0.275	0.364	-0.342
Year 3 (only winter)	-0.236	0.168	0.240	0.206	-0.064	1.535	-	-	-	-	-
Year 4 (only winter)	0.030	0.337	0.362	0.638	0.138	0.532	-	-	-	-	-
nds_b100	NA	3.320	NA	NA	NA	NA	NA	NA	NA	NA	10.664
(nds_b100)^2	NA	-5.733	NA	NA	NA	NA	NA	NA	NA	NA	NA
icw_b100	NA	-2.150	NA	NA	NA	NA	NA	NA	NA	NA	NA
ncs_b100	NA	NA	NA	NA	4.909	NA	NA	NA	3.083	1.744	NA
(ncs_b100)^2	NA	NA	NA	NA	-2.825	NA	NA	NA	NA	NA	NA
ndw_b100	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
ncs_r1000	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	0.078
nds_r1000	NA	NA	NA	0.029	NA	NA	NA	NA	NA	NA	NA
ics_r1000	NA	NA	NA	NA	NA	NA	NA	1.889	NA	NA	NA
(ics_r1000)^2	NA	NA	NA	NA	NA	NA	NA	-0.026	NA	NA	NA
ndw_r1000	NA	NA	NA	NA	0.291	NA	NA	NA	NA	NA	NA
wbs_r1000	NA	NA	NA	NA	NA	NA	0.0557	NA	NA	NA	NA
(wbs_r1000)^2	NA	NA	NA	NA	NA	NA	-3.985e-04	NA	NA	NA	NA
dcsal	NA	NA	NA	NA	NA	NA	2.017e-04	NA	NA	NA	NA
dcan	NA	NA	NA	NA	NA	-5.584e-04	NA	NA	NA	NA	NA
dcwet	NA	NA	NA	NA	NA	NA	NA	0.0012	-3.69e-04	NA	NA
dcs_r1000	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
dmmi	NA	NA	NA	-7.14e-04	NA	NA	NA	NA	3.555e-04	NA	NA
(dmmi)^2	NA	NA	NA	0.122	NA	NA	NA	NA	NA	NA	NA
% Explained deviance	20.41	39.92	9.16	57.42	81.80	44.39 *N.S. null model	51.25	44.16 *N.S. null model	89.53	36.98	57.76 *N.S. null model

Conclusiones finales

Conclusiones

Las principales conclusiones obtenidas a lo largo de este trabajo se presentan en dos grupos: (1) conclusiones generales descritas desde una perspectiva puramente académica (científico-técnica) y (2) recomendaciones y directrices de gestión y manejo que puedan ser aplicadas a través de diferentes instrumentos de gestión a diferentes escalas, en complejos de humedales costeros como el Mar Menor y otros análogos.

Principales conclusiones científicas

- ✓ La comunidad de aves acuáticas, una de las taxocenosis de mayor valor biológico y funcional, se ha visto notablemente modificada durante las últimas décadas. La intensificación agrícola, el turismo y las actividades relacionadas, han sido las principales fuerzas motrices que han llevado al Mar Menor y sus humedales asociados a un profundo cambio ambiental que ha repercutido en su biota.

- ✓ Los factores locales de presión antrópica derivados de dichas fuerzas motrices (drenajes agrícolas, alteración estructural por urbanismo, etc.) provocan procesos de degradación ecológica (por ejemplo, la eutrofización de las aguas). La respuesta de los humedales, en distintas escalas espaciales (desde pequeñas charcas hasta todo el complejo de humedales) puede describirse como una sucesión de fases caracterizadas por sus comunidades ornitológicas y por cambios fisicoquímicos claramente identificables.

- ✓ Dichas fases de cambio se caracterizan, desde un punto de vista biológico general y en particular para la avifauna acuática, por una sustitución progresiva de taxones especialistas (normalmente de alto interés para la conservación) por taxones generalistas y oportunistas (por lo general, de menor valor conservacionista), lo que se traduce en una banalización de la comunidad, manifestada por una pérdida general de riqueza y diversidad.

- ✓ Dependiendo de la escala del compartimento afectado y los procesos a los que se ve sometido, pueden producirse incrementos transitorios de riqueza o diversidad ornitológica, promovidos por el aumento de la productividad del sistema y mantenidos temporalmente por ciertos mecanismos de control interno de los procesos de degradación ecológica, que permiten la coexistencia de un mayor número de especies. No obstante, a largo plazo siempre se va a producir una pérdida de singularidad y de valor de conservación general.

✓ A lo largo de esta trayectoria ciertas especies se erigen como indicadores de cambio del estado ecológico de la laguna. Los somormujos y zampullines (*Podicipedidae*) se identifican como indicadores de “alerta temprana” frente al proceso de eutrofización (más claramente en el caso del somormujo lavanco *Podiceps cristatus* que en el del zampullín cuellinegro *P. nigricollis*). La focha común *Fulica atra* se identifica como un indicador de estados más avanzados de eutrofización. El cormorán grande *Phalacrocorax carbo* parece mostrar una respuesta “de fondo” al proceso, no obstante su dinámica local parece más dependiente de otros factores (tendencias biogeográficas, actividad pesquera).

✓ La respuesta de las especies a procesos locales (eutrofización) se conjuga con las preferencias más generales en cuanto a la estructura del paisaje lagunar y terrestre inmediato, la geomorfología de la cubeta y la existencia de determinados elementos (islas, zonas de refugio, puertos, etc.), todo ello mediado por las diferentes estrategias de búsqueda de alimento y requerimientos tróficos de las especies.

✓ La franja de agua más somera y cercana a la línea de costa alberga una elevada diversidad de especies, dado que ofrece una amplia variedad de nichos tróficos que permiten la coexistencia de diferentes morfotipos y estrategias de alimentación. El efecto de la distancia a la orilla es notable para casi todas las especies, tomando signo distinto en función de dichas características ecomorfológicas.

✓ Las áreas naturales ribereñas (paisajes de matorral originales de la orilla de la laguna) con gradientes más acusados (continental-lagunar y lagunar-oceánico) muestran los mayores valores de riqueza, diversidad y uso de aves acuáticas. El somormujo lavanco, la garceta común *Egretta garzetta* y la gaviota picofina *Larus genei* responden positivamente a la presencia de hábitats terrestres naturales en la orilla. Por otro lado, la focha común muestra una respuesta positiva significativa a los cultivos de regadío, lo que es coherente con su papel bioindicador de eutrofización avanzada.

- ✓ La estacionalidad tiene un efecto importante en los índices descritos, siendo el invierno la época de mayor valor general de la comunidad de aves acuáticas (más abundante y diversa). No obstante, durante el verano (cuando las actividades turísticas dominan la laguna), destaca el papel de las áreas protegidas y otros humedales adyacentes a la laguna como lugares de alimentación, reposo y nidificación.

Recomendaciones para la gestión

- ✓ La monitorización estandarizada y regular de las comunidades ornitológicas (e idealmente, de las condiciones limnológicas y otras taxocenosis palustres), junto a una mejor identificación y conocimiento de la secuencia característica de fases de cambio ecológico (en respuesta a procesos como la eutrofización, alteración del régimen hidrológico, etc., y asociada a cambios composicionales, estructurales y funcionales de las taxocenosis), proporcionaría una herramienta efectiva y de bajo coste para la detección, prevención o corrección de cambios no deseables en este tipo de humedales.

- ✓ Frente al problema específico de armonizar el tratamiento de drenajes de origen agrícola o urbano-turístico con la conservación de los ecosistemas naturales en la cuenca vertiente y en la cubeta del Mar Menor, sería recomendable implementar un Sistema de Gestión Local (SGL) basado en crear una red artificial de pequeños humedales (unidades funcionales) compuestas dos charcas con funciones claramente diferenciadas: (a) filtro verde y (b) recuperación de hábitats y aumento de biodiversidad.

- ✓ Acoplado al anterior sería aconsejable desarrollar un Sistema de Gestión Integrado (SGI) que se implementaría a escala de paisaje. Consistiría en una gestión conjunta e integrada de los diferentes canales de drenaje para concentrar y redirigir los efluentes hacia las unidades funcionales del SGL, promoviendo un aumento de los tiempos de retención del agua para un mayor eficiencia depuradora.

- ✓ Se debería promover la recuperación y restauración de los hábitats terrestres naturales (estepa salina, saladar) y los hábitats de orilla, prestando especial atención al mantenimiento de su continuidad espacial y heterogeneidad estructural, así como a la conservación de las “ventanas naturales” a la laguna, potenciando las sinergias positivas laguna-hábitats terrestres para la conservación de sus respectivas comunidades biológicas.

- ✓ Es necesario regular de manera más efectiva, sobre todo durante la época estival, el tipo de actividades humanas (favorecer actividades de bajo impacto y poco estrés) y las zonas donde se realicen, limitándolas, de forma específica en las franjas de orillas naturales, y de forma más general, en la franja de agua más cercana a la línea de costa (alrededor de los primeros 100 metros de agua). Esta regulación podría lograrse a través del uso de ventanas artificiales que concentren las infraestructuras e instalaciones más estresantes y de mayor impacto físico, dejando para el resto de la masa de agua las actividades extensivas más compatibles con la actividad de las aves.

- ✓ Este enfoque podría desarrollarse a través de los planes de gestión que este tipo de sistemas requieren por su protección internacional, en particular en el ámbito de la Unión Europea, y que en la Región de Murcia se están redactando con un enfoque integral de ámbito comarcal. Estos planes de gestión integral deben permitir armonizar los objetivos de conservación de la biodiversidad (Directiva Aves y Directiva Hábitats) con los objetivos de conservación y mantenimiento de la calidad de las aguas (DMA).

- ✓ Parece necesario comenzar a integrar, en las evaluaciones de los humedales en general (y en particular, en los sistemas costeros semiconfinados) el análisis del papel de las aves acuáticas como indicadores ecológicos del estado del sistema, aplicándolos de forma combinada con otros indicadores de calidad y biodiversidad basados en las biocenosis lagunares sumergidas y en los hábitats terrestres asociados a la cubeta.

Conclusions

The main conclusions obtained throughout this work are presented in two groups: (1) general conclusions described from an academic perspective (scientific and technical) and (2) management guidelines and recommendations that can be applied through various management tools at different scales, in coastal wetland complexes like the Mar Menor and analogous sites.

Major scientific conclusions

- ✓ The waterbird community, one of the taxocenoses with the most biological and functional value has been significantly modified over the past decades. Agricultural intensification, tourism and related activities have been the main driving forces have led the Mar Menor and its associated wetlands to profound environmental change that has affected its biota.

- ✓ Local anthropogenic pressure factors derived from these driving forces (agricultural drainage, structural alteration by urbanization, etc.) cause ecological degradation processes (e.g. eutrophication). The response of wetlands, at different spatial scales (from small ponds to the entire wetland complex) can be described as a succession of clearly recognizable phases characterized by their ornithological communities and by easily identifiable physicochemical changes.

- ✓ These phases of change are characterized, from a general biological point of view particularly for waterbirds, by a gradual replacement of specialist species (usually of high conservation concern) by generalist and opportunistic ones (usually of lower conservation value), resulting in a trivialization of the community, expressed by a general loss of richness and diversity.

- ✓ Depending on the scale of the affected compartment and the processes to which it is subjected, transitional increases in ornithological richness or diversity can occur, promoted by an enhancement of the productivity of the system and temporally maintained by certain mechanisms of internal control of ecological degradation processes, allowing the coexistence of a larger number of species. However, in the long term it will always result in a loss of singularity and general conservation value.

- ✓ Along this path, certain species stand out as indicators of change in the ecological status of the lagoon. The grebes (*Podicipedidae*) are identified as indicators of "early warning" of the process of eutrophication (most clearly in the case of the Great Crested Grebe *Podiceps cristatus* than in the Black-necked Grebe *P. nigricollis*). The coot *Fulica atra* is recognized as an indicator of more advanced stages of eutrophication. The Great Cormorant *Phalacrocorax carbo* seems to show a "background response” to the process, although its local dynamics seems more dependent on other factors (biogeographical trends, fisheries).

- ✓ The response of species to local processes (eutrophication) is combined with more general preferences regarding the structure of the immediate lagoon and terrestrial landscape, the geomorphology of the basin and the existence of particular key elements (islands, refuge areas, hasbours, etc..), all mediated by the different foraging strategies and trophic requirements of the species.

- ✓ The shallowest and closest to the shoreline water strip has a high diversity of species, since it offers a wide range of trophic niches that allow the coexistence of different morphotypes and feeding strategies. The effect of the distance to shore is remarkable for almost all species, and acquires different sign depending on such ecomorphological characteristics.

- ✓ The natural shoreline areas (original scrub landscapes of the shore of the lagoon) with steepest gradients (continental-lagoon and lagoon-oceanic) show the highest values for richness, diversity and waterbird use. TheG crested grebe, Little Egret *Egretta garzetta* and Slender-billed Gull *Larus genei* respond positively to the presence of natural terrestrial habitats on the shore. On the other hand, Common coot shows a significant positive response to irrigated crops, which is consistent with its bioindicator role of advanced eutrophication.

- ✓ Seasonality has a significant effect on the indices used to describe the community, winter being the season of greatest overall value of the waterbird community (more abundant and diverse). However, during the summer (when tourism activities dominate the lagoon), the role of protected areas and other wetlands adjacent to the lagoon as feeding, resting and nesting places, is highlighted.

Recommendations for management

- ✓ The standardized and regular monitoring of the ornithological community (and ideally, of the limnological conditions and other lagoon taxocenoses), together with a better identification and understanding of the characteristic sequence of stages of ecological change (in response to processes such as eutrophication, hydrological regime alterations, etc., and associated to compositional, structural and functional changes of taxocenoses), provide a cost-effective tool for the detection, prevention or correction of undesirable changes in these wetlands.
- ✓ Facing the problem of harmonizing the treatment of agricultural or urban-tourism originated drainages with the conservation of natural ecosystems in the watershed and basin of the Mar Menor, it would be advisable to implement a Local Management System (LMS) based on the creation of a network of small artificial wetlands (functional units) composed of two pools with distinct functions: (a) green filter and (b) recovery of habitats and biodiversity enhancement.
- ✓ Coupled to the above described system, it would be advisable to develop an Integrated Management System (IMS), to be implemented at the landscape scale. It would consist in a joint and integrated management of different drainage channels for concentrating and redirecting the effluent to the functional units of the LMS, promoting an increase in water retention times for enhanced depuration efficiency.
- ✓ It should be promoted the recovery and restoration of natural terrestrial habitats (saline steppe, salt marsh) and shoreline habitats, with particular

attention to maintaining their structural heterogeneity and spatial continuity, as well as the conservation of "natural windows" to the lagoon, promoting positive synergies between the lagoon and its surrounding terrestrial habitats for the conservation of their biological communities.

- ✓ It is necessary to regulate more effectively, especially during the summer season, the type of human activities (favoring those with the lowest impact and stress), and limiting them specifically on the strips of natural banks, and more generally, in the water strip closest to the coastline (around the first 100 meters of water). This regulation could be achieved through the use of artificial windows to concentrate the most stressful infrastructure and facilities with greater physical impact, leaving the remainder of the water body open to extensive activities more compatible with waterbird activity.

- ✓ This approach could be developed through the management plans that such systems require due to its international protection, in particular in the European Union, and that in the Region of Murcia are being drafted with an integrated approach on a district scale. These comprehensive management plans should enable to harmonize the objectives of biodiversity conservation (Birds and Habitats Directives) with those of conservation and maintenance of water quality (DMA).

- ✓ It seems necessary to start the integration, in ecological assessments of wetlands in general (and in particular in semi-enclosed coastal systems), of the analysis of the role of waterbirds as ecological indicators of the system’s state, applying them in combination with other quality and diversity indicators based on the lagoon’s submerged biocoenoses and on the terrestrial habitats associated to the basin.