
**Conservation biology of the critically endangered
Balearic shearwater *Puffinus mauretanicus*:
bridging the gaps between breeding colonies
and marine foraging grounds**



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TESI DOCTORAL

Conservation biology of the critically endangered
Balearic shearwater *Puffinus mauretanicus*: bridging the gaps between
breeding colonies and marine foraging grounds

Memòria presentada per Maite Louzao Arsuaga per optar al títol de Doctora del Programa de Doctorat de Ciències Marines del IMEDEA, Universitat de les Illes Balears, sota la direcció dels Drs. Daniel Oro de Rivas i José Manuel Arcos Pros, i ponència del Dr. Rafael Bosch Zaragoza.

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Eskerrak, agradecimientos, agraïments

Ama hizkuntza euskara dudanez harekin hasiko naiz. Ene bizitzako puntu honetara iritsi baino lehenago leku askotatik pasa naiz eta jende askorekin topatu ere. Guztietaz gogoratzen saiatu naiz, baino lortu ez badut eta norbaitekin ahaztu banaiz barkamena aldeaz aurretik eska nahi nioke. Historiak erakusten digun bezala, gaur egun gu geu halakoak bagara iragan eta bizi-esperientzia jakin batzuen emaitza garelako.

Xa que a miña lingua nai é o euskara comezarei con ela. Antes de chegar a este punto da miña vida pasei por moitos sitios e topei con moita xente. Intentei lembralos a todos pero se non o conseguín e me esquecín de alguén querería pedir desculpas por adiantado. Tal e como nos ensina a historia, no presente somos dun determinado xeito porque somos o resultado dun pasado e dunhas experiencias de vida determinadas.

Ja que la meva llengua mare és l'euskera començaré amb ella. Abans d'arribar en aquest moment de la meva vida he passat per molts de llocs i m'he trobat amb molta gent. He intentant enrecordar-me'n de tots però sino ho he aconseguit i me n'he oblidat d'algú m'agradaria demanar les meves disculpes per endavant. Com ens mostra la història, al present noltros mateixos som així perquè som el resultat d'un passat i experiències de vida determinades.

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AMOMIAK: Ipurdi handi kargadunak. Olatz, Marta eta Ainhoa. “*Bailara batean toki ezaguna, euskararen deia izan da entzuna, Euskal Herri osoan sendo erantzuna, Oiartzunen euskarak badu oihartzuna*”. Nahiz eta distantziak hainbeste urteetan bananduak izan gaituen, nerabegarotik nire ondoan egon zarete une oro. Mila esker.

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Os comezos na investigación. Ai Pilotxa, eses comezos mostreando na lonxa de Vigo, medindo ollos e contando ampollas de Lorenzini dos marraxos. ¿Cómo se facía iso da R² no Excel? ¡¡¡¡Vaia peste camiño da Facultade!!!! ¿E o primeiro contacto coa Dinámica Sedimentaria? ¡Ai! Sempre lembro a zona intermareal de Ramallosa, medindo *ripples*, *megaripples* e máis formas de fondo!!! “*¡Á Ramallosa, por ir ao baile morena perdín a moza, perdín a moza, deixala ire! No canto desa morena outra ha de vire...*” ¿E esa colaboración coa Irene e Javi de Xeociencias? E esas miradas telepáticas que lles enviabamos aos teus pais (Maribel e Manolo) dende a frecha areosa de Arealonga, ¿eh Irelou?

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Maite, Maitetxu, Tetxu, Tetxulina, Maripertxenta, Triki

Capítulo 1



Introducción

1.1 ¿Qué es la biología de la conservación?

La biología de la conservación se refiere a la conservación de todos los componentes de la biodiversidad (genes, poblaciones, especies, hábitats, comunidades y ecosistemas) que son perturbados directa o indirectamente por actividades humanas u otras, así como también a las interacciones entre dichos componentes con el medio físico que los rodea (Soulé 1985, Meffe y Carroll 1994, Fazey et al. 2005).

La biología de la conservación es una ciencia multidisciplinar que surgió como respuesta de la comunidad científica a la actual crisis de biodiversidad y a la escala trepidante de cambios ambientales (Soulé 1985, Orians y Soulé 2001). Se basa en los principios de la ecología, biogeografía, genética de poblaciones, economía, sociología, antropología, filosofía, y otras disciplinas con base teórica para preservar la diversidad biológica del planeta (Meffe y Carroll 1994). Su principal objetivo es proporcionar un marco teórico y desarrollar las herramientas técnicas que permitan a la sociedad en su conjunto anticiparse, prevenir y reducir los daños ecológicos y generar información científica sobre la cual diseñar e implementar políticas de conservación efectivas (Orians y Soulé 2001). Anticiparse porque la conservación necesita que del análisis reactivo actual (el análisis derivado una vez la problemática de conservación se ha detectado) pasemos a una ciencia activa que nos permita anticiparnos a las crisis en

desarrollo y preparar planes de contingencia con una base científica (Brooks et al. 2006). Las ciencias no biológicas, como la economía, ciencias sociales y políticas, son importantes también ya que, realmente, tendrían un mayor impacto en la conservación de la biodiversidad que las propias ciencias biológicas (Meffe y Carroll 1994).

La biología de la conservación se basa en tres principios básicos (ver Meffe y Carroll 1994): el primero nos dice que la evolución une toda la biología, ya que ésta es capaz de explicar los patrones de diversidad actual y ofrece una perspectiva histórica de la dinámica de la vida. El segundo nos señala que los sistemas ecológicos son dinámicos y no se encuentran en equilibrio. Los ecosistemas son sistemas abiertos con flujos de especies, materiales y energía y deben de ser entendidos en el contexto del medio que los rodea. Y por último, que la presencia humana se debe incluir en los planes de conservación, al ser imposible proteger la naturaleza de la influencia humana ya que las reservas naturales se encuentran rodeadas por sistemas altamente explotados y es imposible aislarlas de las influencias externas. Al mismo tiempo, integrar a la población local en las áreas protegidas es importante, en primer lugar porque su conocimiento de la historia natural de la zona puede ser de gran ayuda a la hora del diseño de los planes de gestión de las reservas, y en segundo lugar porque prescindir de ella puede dar lugar a conflictos.

1.2 La importancia de la biología de la conservación en el medio marino

La biología de la conservación es una ciencia reciente que comenzó a desarrollarse hace dos décadas, centrado principalmente en estudios terrestres (Norse y Crowder 2005). El ecosistema marino es el gran olvidado de esta ciencia ya que hasta 1986 no se publicó el primer capítulo sobre conservación marina (Johannes y Hatcher 1986) y aún en el periodo 1997-2003 solamente el 10% de las publicaciones trataban esta temática (Levin y Kochin 2004). Todo ello refleja que no sólo el mar es un gran desconocido para la biología de la conservación, sino que la conservación de la biodiversidad marina es pasada por alto incluso por los científicos marinos (Norse y Crowder 2005). La biodiversidad marina es más extensa y vulnerable de lo que se pensaba y así, la biología de la conservación se presenta como una ciencia clave para determinar los factores y causas del declive de importantes taxones y ecosistemas, proporcionando herramientas y medidas de conservación para contrarrestar su estado actual (Norse y Crowder 2005, Roberts 2005, Hendriks et al. 2006).

Aunque la vasta riqueza biológica de los océanos esté aún por descubrirse (Hendriks et al. 2006), los cambios ambientales producidos por el ser humano ya han tenido un efecto negativo sobre el funcionamiento de los ecosistemas y la dinámica de población de muchos organismos (Botsford et al. 1997, Hutchings 2000, Worm et al. 2005, UNEP 2006). Por ejemplo, la pesca, una de las actividades antropogénicas más amenazadoras para el ecosistema marino (Jackson et al. 2001, Heppel et al. 2005, Pauly et al. 2005), es responsable del declive de especies marinas apicales (90% de los grandes peces depredadores) y de la reducción en un 80% de la biomasa de la comunidad (Myers y Worm 2003, Lewison et al. 2004). Existen evidencias que indican que las cadenas tróficas marinas están conectadas a escalas oceánicas (Hendriks et al. 2006), pero las fuerzas que dirigen estas conexiones se conocen poco. Es, por tanto, imprescindible mejorar nuestro conocimiento sobre el ecosistema oceánico global para diseñar redes de áreas protegidas que preserven la biodiversidad de una forma efectiva (Leslie 2005, UNEP 2006).

En los últimos años, se está consiguiendo que las estrategias de conservación marina y gestión pesquera integren todos los componentes del ecosistema en su conjunto (Pikitch et al. 2004, Norse et al. 2005). A escala planetaria se están utilizando redes de áreas marinas protegidas para gestionar pesquerías y proteger especies y hábitats marinos amenazados (Gerber et al. 2005, Roberts 2005). Las especies con una gran capacidad migradora, como aves marinas, cetáceos, tortugas marinas y grandes peces pelágicos, centran un mayor interés en la actualidad, ya que hasta ahora se había puesto un mayor esfuerzo en la protección de especies sésiles o con poca movilidad a la hora de diseñar reservas marinas (Norse et al. 2005). Estas especies más móviles son especialmente susceptibles cuando se agregan en altas densidades en el tiempo (durante la reproducción) y espacio (áreas de alimentación) y requieren del diseño de reservas más complejas (Anderson et al. 2003, James et al. 2005, Norse et al. 2005, Hyrenbach et al. 2006).

1.3 Rasgos de historia de vida de las aves marinas: implicaciones para su conservación

Las aves marinas son organismos longevos caracterizados por una alta supervivencia y bajas tasas de reproducción (Weimerskirch 2002). La supervivencia y la reproducción son rasgos de historia de vida fundamentales, ya que la combinación de ambos nos da la tasa a la cual una población aumenta o decrece, aunque también hay que tener en cuenta otros parámetros demográficos como el reclutamiento o la emigración (Weimerskirch 2002). Su historia de vida específica hace que las aves marinas sean organismos vulnerables al no poder responder de forma rápida a la reducción de sus densidades poblacionales, debido a sus altos tiempos de generación (Croxall y Rothery 1991, Musick 1999). La supervivencia adulta es el parámetro demográfico que más influye en la tasa de crecimiento poblacional de un organismo longevo a nivel local, mientras que la emigración y la inmigración lo son a nivel metapoblacional (Croxall y Rothery 1991, Oro 2003).

El comportamiento adaptativo es también un aspecto importante en la dinámica de población de las aves marinas (Russell 1999). La teoría de la historia de vida predice que las aves marinas deben tomar decisiones en cuanto a sus estrategias reproductivas a lo largo de sus vidas (Stearns 1992). Estas decisiones se refieren a cómo distribuir los recursos entre la reproducción presente frente a la propia supervivencia y futura reproducción, debido a los costes ecológicos y fisiológicos asociados a la cría (Russell 1999). En un escenario donde los recursos estén limitados, reproducirse en un momento dado puede influir negativamente en la probabilidad de supervivencia hasta la siguiente reproducción, y una especie longeva debe limitar el riesgo (a morir), sobre todo, durante los primeros años de vida, para poder incrementar su éxito reproductor global (Weimerskirch 2002).

El estudio de la demografía es determinante para entender la evolución de la historia de vida, ya que permite examinar la intensidad de la selección sobre los rasgos vitales (Krebs 2001, Weimerskirch 2002). Los modelos demográficos son una herramienta imprescindible en la biología de la conservación, ya que nos ayudan a (1) evaluar el estado de conservación de una población, y en caso de detectar problemas (2) diagnosticar las causas (diagnóstico), (3) determinar la fase del ciclo de vida más adecuada en la cual poder llevar a cabo una intervención (prescripción) y (4) predecir el destino más probable de la población dadas una determinada evaluación, diagnóstico

y prescripción (prognosis) (Caswell 2001). Así, un estudio demográfico nos permite realizar predicciones sobre la vulnerabilidad y formular estrategias de conservación (Musick 1999).

Actualmente todos los ecosistemas naturales en el planeta han sido alterados por el ser humano (Meffe y Carroll 1994, Brooks et al. 2006). Un número enorme de especies se ha extinguido, los ciclos hidrológicos y químicos se han modificado, se ha perdido una gran extensión de cobertura vegetal, la diversidad genética se ha degradado y el clima del planeta ha sido alterado (Meffe y Carroll 1994, Jenkins et al. 2003, Stocker y Raible 2005, Malcolm et al. 2006). Una vez se ha constatado la existencia de un problema de conservación, la diagnosis es imprescindible para identificar los parámetros demográficos que afectan negativamente a la tasa de crecimiento poblacional de un organismo amenazado (Krebs 2001, Norris 2004). La diagnosis de conservación se basa en comparar los rasgos de historia de vida a diferentes escalas espacio-temporales (Caswell 2001), dentro de la misma especie (entre diferentes poblaciones) o al menos entre especies filogenéticamente relacionadas y con historias de vida similares (Barbraud et al. 1999, Dhondt 2001). Esto es debido a que los valores de los parámetros demográficos son el resultado de las adaptaciones individuales y las condiciones ambientales que los rodean (Weimerskirch 2002). Por tanto, la comparación de los rasgos de historia de vida permiten determinar la existencia de una variabilidad espacio-temporal y sus causas y así poder entender, en último término, la evolución de los rasgos de historia de vida (Weimerskirch 2002).

En la naturaleza la mayoría de las especies se constituyen en múltiples poblaciones, es decir, metapoblaciones (Harrison 1994) cuya estructuración espacial y dinámica tienen implicaciones en la especie en su conjunto y están conectadas por la migración (Hanski 1998). La estructura espacial de las interacciones ecológicas también afecta a las poblaciones tanto como las tasas de fecundidad y mortalidad, la competencia y la depredación (Hanski 1998, Lipcius et al. 2005). La teoría de metapoblaciones es importante en los estudios de conservación para entender las amenazas a las que se enfrentan las especies a una escala espacial razonable sobre la que aplicar medidas de conservación (Harrison 1994, Lipcius et al. 2005).

1.4 Las aves marinas en un ecosistema modificado: implicaciones para su conservación

La actividad antropogénica de los últimos siglos ha afectado especialmente a los organismos de niveles tróficos superiores, como las aves marinas, ya que en estos niveles las alteraciones del ambiente se ven amplificadas (Krebs 2001). En el caso de las aves marinas, sus rasgos de historia de vida (tasas de supervivencia, condición adulta, crecimiento juvenil, edad de la primera reproducción, etc.) se ven enormemente afectados por las modificaciones que se realizan en el ecosistema marino, al pasar más del 90% de su tiempo de vida y dos terceras partes de su ciclo anual en el mar (Furness y Camphuysen 1997, Heppel et al. 2005). Estos impactos antropogénicos están provocando cambios importantes en las densidades poblacionales de las aves marinas como resultado de la interacción de diversos factores, afectando sus rasgos de historia de vida y sus dinámicas poblacionales, así como la estructura de las comunidades (ver revisión en Montevecchi 2002, Oro 1996, Furness 2003, Lewison et al. 2004, Votier et al. 2004, Arcos et al. 2006).

Las estrategias de conservación deben de comenzar por determinar y entender las mayores amenazas a las que se enfrentan las especies (Boersma et al. 2002). La modificación del hábitat es la causa más frecuente por la cual una especie entra en la categoría de amenazada (Meffe y Carroll 1994). El hábitat de las aves marinas se podría dividir en el hábitat de nidificación y alimentación, tanto durante el periodo reproductor como no reproductor (Boersma y Parrish 1998).

El hábitat de nidificación

El hábitat de nidificación se ha visto modificado por la extracción de guano en los últimos 200 años, pero la destrucción y modificación masiva ha ocurrido en los últimos 50-70 años debido, principalmente, al crecimiento demográfico y desarrollo urbanístico en la zona costera (Boersma et al. 2002). La introducción de especies en las colonias de aves marinas es una de las causas principales del declive de muchas especies (Burger y Gochfeld 1994, Czech et al. 2000). Las especies depredadoras (pequeños carnívoros principalmente) son especialmente perjudiciales ya que muchas poblaciones no presentan depredadores naturales y las aves marinas no han desarrollado estrategias apropiadas de defensa en cuanto a comportamiento, ecología y reproducción (Loope y

Mueller-Dombois 1989). Las campañas de erradicación son necesarias para eliminar depredadores introducidos, planeándolas y ejecutándolas cuidadosamente para evitar efectos adversos sobre la especie en cuestión (Burger y Gochfeld 1994).

La causa más importante del declive de las aves marinas en los últimos 200 años ha sido la explotación humana (Boersma et al. 2002 y referencias) y los reducidos tamaños poblacionales actuales son su consecuencia directa y/o indirecta. La caza para alimento, plumas ornamentales y aceite, y la recolección de huevos fueron históricamente las causas principales, aunque la explotación sigue siendo importante en localidades específicas (Burger y Gochfeld 1994). Las molestias debidas a la presencia humana en las colonias, tanto relacionadas con el ecoturismo como la investigación, provocan el abandono temporal y deserción de los nidos y aumentan el riesgo de depredación, pudiendo llegar a reducir el éxito reproductor (Burger y Gochfeld 1993). Reducir los efectos de la presencia humana es sencillo si se llevan a cabo medidas de gestión adecuadas (Boersma et al. 2002 y referencias).

El hábitat de alimentación

Los factores que más afectan al hábitat de alimentación de las aves marinas son la pesca y la contaminación, aún sin restar importancia al cambio climático (Jackson et al. 2001, Pauly et al. 2002, Boersma et al. 2002, Heppel et al. 2005, Worm et al. 2005). En ausencia de actividades antropogénicas, las poblaciones de aves marinas y sus presas fluctúan de forma estocástica inducidas por cambios climáticos y oceanográficos, así como por fluctuaciones poblacionales intrínsecas (Carscadden et al. 2001). Tanto los cambios ambientales naturales como las actividades humanas pueden inducir cambios en la estructura de la comunidad a gran escala y, a su vez, interactuar entre ellos (Steele 1998). Por tanto, es difícil, a veces, diferenciar los factores naturales de los antropogénicos que perturban y provocan cambios en el ecosistema marino (Nisbet 1994, Tasker et al. 2000).

La mortalidad en artes de pesca, principalmente en el palangre, es una de las mayores amenazas para las aves marinas (ver revisión en Montevecchi 2002), ya que su historia de vida las hace particularmente sensibles a los factores que aumentan la mortalidad adulta (Croxall y Rothery 1991, Russell 1999). Así, la captura incidental en el palangre ha llevado a muchas especies de albatros y petreles al declive crónico y/o extinción (Tuck et al. 2001, Lewison et al. 2004) y provoca la muerte de cientos de miles

o incluso millones de aves anualmente (Brothers et al. 1999, Cooper et al. 2003). Más de una tercera parte de las especies que se ven afectadas por esta interacción están catalogadas como amenazadas según los criterios IUCN (Unión Internacional para la Conservación de la Naturaleza) (Brothers et al. 1999, Furness 2003). Casi toda la información disponible proviene de pesquerías reguladas y se desconoce la importancia de este impacto en pesquerías no reguladas (Brothers et al. 1999), aunque es esperable que sea aún mayor.

Otras interacciones incluyen la competencia, ya que las aves marinas y la pesca explotan los mismos recursos marinos o interrelacionados (principalmente recursos pelágicos). Así, las pesquerías reducen la abundancia y disponibilidad de las presas de las aves marinas, incluso provocando su colapso, llegando a afectar negativamente la reproducción y densidades de población de las aves marinas (ver revisión en Montevecchi 2002, Jackson et al. 2001, Furness 2003). Por el contrario, determinadas artes de pesca proporcionan un recurso trófico adicional (descartes y despojos) del cual se aprovechan especies con un alto comportamiento adaptativo (Garthe et al. 1996, Furness 1999, Oro 1999, Tasker et al. 2000, Arcos 2001). Este alimento añadido puede afectar los rasgos de historia de vida y dinámica de población de las aves marinas, así como la estructura de la comunidad (Oro 1999, Votier et al. 2004). Este recurso trófico puede llegar a aumentar la reproducción (tasas de ceba de los pollos, tamaño de puesta y huevos, tiempo de cuidado parental) e incluso la supervivencia de las especies implicadas (Oro et al. 1995, Oro y Furness 2002, Oro et al. 2004), favoreciendo potencialmente el incremento de sus densidades poblacionales (Oro 1999, Furness 2003).

En los últimos años se está promoviendo una gestión pesquera que priorice el ecosistema en su conjunto más que las especies objetivo, a través de una explotación sostenible de los recursos vivos marinos (Pikitch et al. 2004). El uso de medidas de mitigación (líneas espantapájaros, calar el aparejo por debajo de la superficie del agua, aumentar la velocidad de hundimiento y evitar los descartes) son imprescindibles para reducir la captura incidental de aves en el palangre (Montevecchi 2002 y referencias) y así poder invertir las tendencias poblacionales actuales de muchas especies. En aplicación de las políticas pesqueras actuales, la reducción de la disponibilidad de descarte podría reducir, a corto plazo, la disponibilidad de alimento de las especies oportunistas obligándolas a buscar recursos tróficos alternativos (Stenhouse y Montevecchi 1999). Esta reducción podría afectar desfavorablemente a la reproducción

y tamaños poblaciones, al igual que a la comunidad de aves marinas en su conjunto (Reeves y Furness 2002). Esta regulación también podría suscitar la competencia intra e inter-específica así como el aumento de la depredación sobre las especies más especializadas y vulnerables (Oro 1996, Stenhouse y Montevecchi 1999, Furness 2003, Votier et al. 2004). Por tanto, es necesario predecir los efectos de las regulaciones pesqueras sobre las poblaciones de aves marinas y tomar medidas preventivas, integrándolas así en la gestión pesquera (Arcos et al. 2006).

En cuanto a la contaminación, las aves marinas están expuestas a un amplio rango de contaminantes al pasar la mayor parte del tiempo en ambientes acuáticos, donde éstos se mueven con mayor rapidez que en ambientes terrestres (Burger y Gochfeld 2002). La contaminación puede tener un efecto difuso (acumulación de metales pesados y organoclorados incorporados a través de la dieta e ingestión de plásticos; Vlietstra et al. 2002) como concentrado con consecuencias desastrosas (vertido de hidrocarburos, ver por ejemplo Peterson et al. 2003) para las aves marinas (ver revisión en Nisbet 1994). Aunque la presencia de metales pesados en las aves marinas puede ser de origen natural, especialmente en especies pelágicas, altas concentraciones pueden llegar a afectar a la reproducción, al igual que los compuestos organoclorados (Nisbet 1994). El efecto de los plásticos es aún incierto, pero puede incluir exposición a contaminantes químicos (Ryan et al. 1988) o el impedimento de la digestión, con la consecuente disminución del éxito reproductor o inanición (Ryan 1987). Tanto la explotación (Wiese et al. 2001) como el transporte (Salomone 2002, Peterson et al. 2003) de hidrocarburos han generado una alta mortalidad en las aves marinas (incluyendo eventos de mortalidad local masiva), tanto reproductoras como migratorias y en zonas costeras como de mar abierto (Heubeck et al. 2003). La contaminación puede llegar a afectar directamente la salud, crecimiento y reproducción de las aves marinas, así como indirectamente a través de las múltiples interacciones bióticas y abióticas del ecosistema (Heubeck et al. 2003, Peterson et al. 2003, Martínez-Abraín et al. 2006). Dada la complejidad de los procesos físicos, químicos y biológicos que determinan la exposición y efectos de los contaminantes las relaciones causa-efecto son difíciles de establecer, especialmente a nivel poblacional (Nisbet 1994).

La magnitud de un impacto determinado dependerá, en gran medida, de la intensidad de la asociación de las aves marinas al hábitat y del grado de agregación de las especies, siendo especialmente vulnerables cuando se concentran en altas densidades (Nel et al. 2000). Cada vez se tiende más a adoptar una aproximación

espacial explícita para atenuar la magnitud de los impactos antropogénicos, por lo que las medidas de protección intensivas de pequeña escala (cierres espacio-temporales y áreas marinas protegidas, por ejemplo) se utilizan para proteger densas agregaciones de aves marinas y hábitats importantes (áreas de muda y alimentación alrededor de colonias, por ejemplo) (Hyrenbach et al. 2000, Anderson et al. 2003). A mayor escala, las regulaciones y medidas de mitigación de captura incidental sirven para atenuar otros impactos dentro del área de campeo de la población (Gilman 2001). Por ejemplo, desviar las rutas de buques petroleros de regiones de alta productividad, como la plataforma continental y el talud, puede ser política y logísticamente poco realista, pero excluir la pesca de pequeñas áreas marinas protegidas cercanas a las colonias de cría y áreas de alimentación sería una medida más factible (Hyrenbach et al. 2000, Anderson et al. 2003).

En un ecosistema modificado, el uso de organismos que proporcionan información sobre la calidad del medio que los rodea es de enorme interés, ya que nos pueden ayudar a detectar los cambios ambientales que se están produciendo en el sistema, tanto a corto como a largo plazo. Las aves marinas son excelentes indicadoras del estado del medio marino (principalmente de la disponibilidad de stocks de especies comerciales y niveles de contaminación; ver revisión en Furness y Camphuysen 1997, Burger y Gochfeld 2002, Velarde et al. 2004). Presentan características adecuadas que las hacen útiles como organismos biomonitores (bioindicadores): su naturaleza colonial permite realizar un seguimiento de las densidades poblacionales sin gran esfuerzo y recoger gran cantidad de datos de un lugar en un periodo de tiempo corto. Los rasgos de historia de vida que responden de forma rápida y sensible a cambios ambientales específicos son los parámetros que se deberían controlar (Russell 1999), como son aquellos relacionados con la alimentación (composición de la dieta, entre otros), en el caso de las aves marinas (Monaghan 1996, Furness y Camphuysen 1997). En cuanto a los contaminantes, éstos se amplifican a través de las cadenas tróficas por lo que las aves marinas son organismos biomonitores (bioindicadores) ideales (Burger y Gochfeld 2002), siendo su efecto más evidente en los embriones, en el éxito de eclosión y desarrollo y comportamiento de los pollos (Furness y Camphuysen 1997). Sin embargo, es imprescindible conocer en detalle la ecología general de las aves marinas, así como tener estimas robustas de la densidad y productividad de muchas poblaciones y especies, ya que se debe conocer el mecanismo causal que provoca una única respuesta

(o limitadas) para poder separar el efecto de determinados cambios ambientales de otros y, así, poder predecirlos (Furness y Greenwood 1993).

1.5 Modelo de estudio

(I) El ecosistema: el mar Mediterráneo occidental

El mar Mediterráneo es un mar semi-cerrado y alargado, con circulación estuarina inversa compuesta por dos cuencas que se encuentran conectadas por el estrecho de Sicilia (Estrada et al. 1985). Existe un exceso en el balance evaporación/precipitación haciendo aumentar la salinidad del Mediterráneo, si la comparamos con el Atlántico (Estrada et al. 1985), y que el flujo saliente sea un 10% más salino que el entrante (Allen et al. 2002). Está caracterizado por un gradiente oligotrófico hacia la cuenca oriental, pudiendo darse dos regímenes tróficos diferentes en invierno y verano (Allen et al. 2002). En invierno, la productividad primaria es alta con un régimen mesotrófico caracterizado por procesos oceanográficos como la mezcla profunda, frentes y afloramientos. En verano, en cambio, el régimen es oligotrófico y diferenciado por una baja productividad primaria donde la estratificación es el fenómeno más característico.

En el Mediterráneo occidental se forman diferentes masas de agua. La circulación superficial de la cuenca occidental, hasta los 200 m de profundidad, consiste en el Agua Atlántica Modificada (MAW) que se origina a partir del agua atlántica superficial entrante y se va modificando progresivamente por la interacción mar-aire y mezclándose a lo largo de la cuenca occidental mediterránea (Send et al. 1999). A profundidades intermedias, entre los 200 y 1000 m, se localizan el Agua Intermedia Levantina (LIW) y el Agua Intermedia Invernal (WIW), generadas en la cuenca oriental y occidental, respectivamente (Send et al. 1999). La masa de agua más característica formada en la cuenca occidental es el Agua Profunda del Mediterráneo Occidental (WMDW): se localiza a una profundidad mayor de 2000 m y se forma por convección profunda en el golfo de León durante el invierno (Estrada et al. 1985, Send et al. 1999).

La circulación superficial de la cuenca occidental corresponde al Agua Atlántica Modificada (MAW) y Agua Intermedia Invernal (WIW) (Milot 1999) (Figura 1.1). En la parte sur de la zona de estudio el agua atlántica entra por el estrecho de Gibraltar, llegando al mar de Alborán, donde describe un giro anticiclónico casi permanente (Milot 1999) (ver Figura 1.2). A continuación, la corriente, llamada comúnmente

‘Almeria-Oran jet’, se desplaza desde la costa ibérica hacia Argelia, caracterizada por la presencia de remolinos (“eddies”) (Millot 1999). Estrada (1996) describió tres procesos hidrográficos que promueven el enriquecimiento en nutrientes de las aguas superficiales en esta zona. La primera sería (1) la mezcla turbulenta en el estrecho que arrastra nutrientes desde el fondo hacia la zona eufótica; la segunda (2) el afloramiento en el mar de Alborán y zonas de frente (como el frente Almería-Orán), asociado a giros anticiclónicos causados por la entrada de agua atlántica y (3) el efecto de la corriente atlántica, que presenta un mayor contenido en nutrientes que las aguas mediterráneas superficiales y que a lo largo de la costa argelina puede producir puntos de alta productividad primaria asociados a meandros y remolinos.

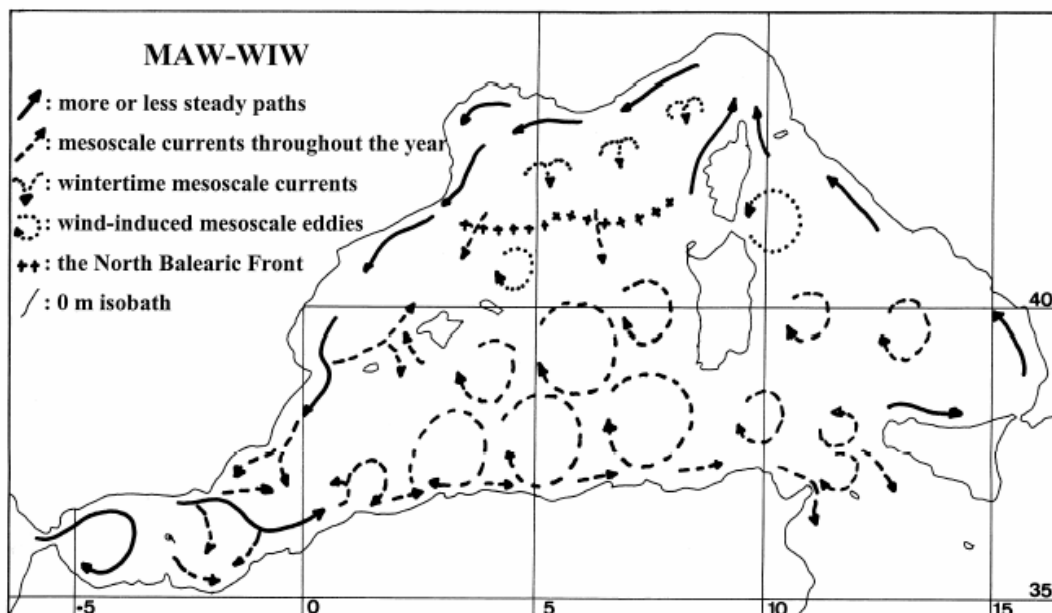


Figura 1.1 Circulación del Agua Atlántica Modificada (MAW) y Agua Intermedia Invernal (WIW) en el Mediterráneo occidental. Los símbolos se explican en la leyenda (adaptado de Millot 1999).

El norte, en cambio, está caracterizado por la presencia de la corriente del Norte (o corriente ‘Liguro-Provenço-Catalan’) que se origina al este del mar de Liguria y se desplaza hacia el sur, por todo el talud continental, hasta el canal de Eivissa (Millot 1999) (Figura 1.1 y Figura 1.2). En la zona del delta del Ebro, se ha identificado la presencia de un frente caracterizado por fuertes gradientes de salinidad, que persiste a lo largo de todo el año y separa las aguas más densas (oceánicas) de las aguas menos densas de origen continental (Rippeth et al. 2002). La posición geográfica y la fuerza del frente limitan la dispersión, hacia alta mar, de larvas de especies costeras y de plataforma (Sabatés 1996). La mezcla de agua inducida por los vientos del noroeste

fuertes, fríos y secos que soplan en el Mediterráneo noroccidental conducen al afloramiento de nutrientes y el consiguiente aumento de la producción planctónica, concretamente en la plataforma del golfo de León y al norte del delta del Ebro en invierno (Salat 1996, Lloret et al. 2001). Por tanto, la productividad del área del delta del Ebro es considerada especialmente alta, dentro de un contexto mediterráneo, debido al aporte de agua dulce del río Ebro, los vientos fuertes y predominantes del noroeste y una plataforma continental relativamente ancha (Salat et al. 2002), unido a la influencia del frente (Estrada 1996, Salat 1996). Las implicaciones biológicas que estos procesos físicos llevan asociados hacen del área del delta del Ebro una zona de alta productividad y favorable de desove para muchas especies de peces, sustentando al mismo tiempo una de las flotas pesqueras más importantes del Mediterráneo occidental (Palomera 1992, Estrada 1996, Salat 1996, Olivar et al. 2001, Agostini y Bakun 2002).

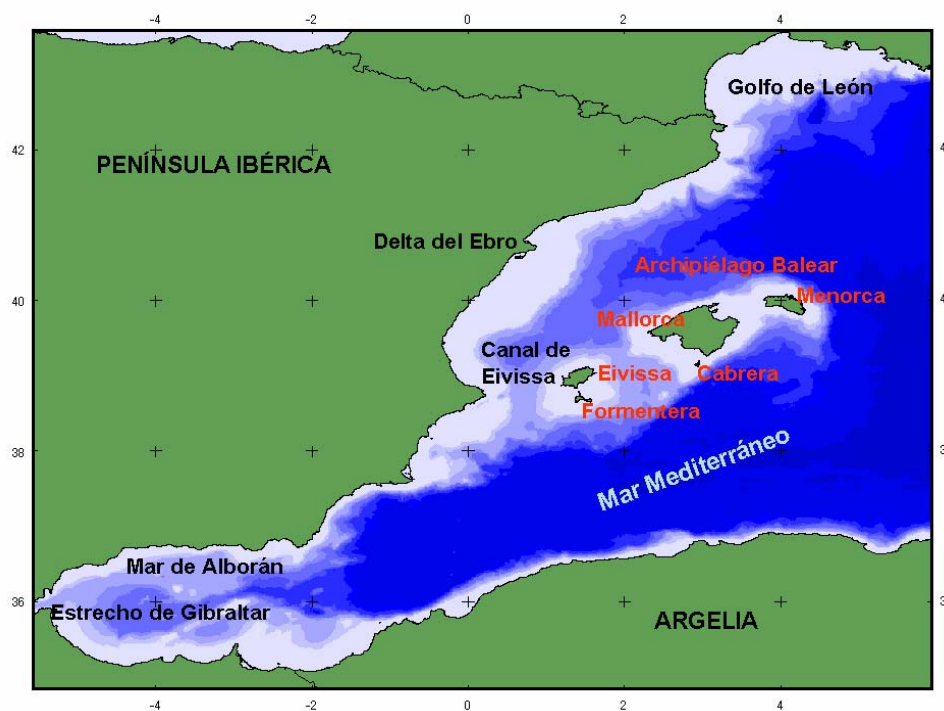


Figura 1.2 El mar Mediterráneo occidental. Se indican las referencias geográficas mencionadas en el texto (en rojo el archipiélago balear). El área de color azul más claro indica la extensión de la plataforma continental (máximo de profundidad de 200 m).

El Mediterráneo occidental está altamente explotado (la mayoría de los stocks demersales están totalmente explotados o sobreexplotados y algunos pelágicos presentan indicios de sobreexplotación) y los niveles actuales de presión pesquera y ambiental pueden llegar a inducir, probablemente, cambios importantes en el

ecosistema (Bianchi y Morri 2000, Tudela 2004, Coll et al. 2006). Sin embargo, se necesitan largas series temporales para profundizar en los análisis y así establecer tendencias poblacionales de los organismos marinos (Papaconstantinou y Farrugio 2000). En esta región biogeográfica, las pesquerías se caracterizan por capturas multiespecíficas y la ausencia de grandes stocks monoespecíficos (Farrugio et al. 1993, Lleonart y Maynou 2003). Recientemente, un modelo ecológico mostró que el ecosistema está altamente condicionado por los depredadores, tanto naturales como antropogénicos, e identificó el arrastre demersal como el arte de pesca con más impacto y menos efectivo, tanto en especies objetivo como no objetivo (Coll et al. 2006).

La gestión pesquera no ha prestado aún suficiente atención a otros componentes del ecosistema que no sean las especies objetivo y no se han controlado las capturas incidentales, ni restringido las artes de pesca y distribuido espacio-temporalmente las pesquerías y áreas marinas protegidas, entre otras (Papaconstantinou y Farrugio 2000, Tudela 2004). Las regulaciones actuales se basan en el control del esfuerzo, en establecer tallas mínimas de captura y de luz de malla de las redes. En algunos casos, como en el levante Ibérico, se establecen cierres espacio-temporales, aunque los espaciales no siempre se respetan y los criterios biológicos para implementar los temporales no siempre están claros (ver más detalles en Lleonart y Maynou 2003). Sin embargo, las pesquerías mediterráneas son difíciles de regular debido a su complejidad y diversidad, así como a las presiones socio-políticas (Lleonart y Maynou 2003).

El mar Mediterráneo está caracterizado por un alto grado de endemismo, incluyendo una comunidad reproductora de aves marinas con tamaños poblaciones limitados (Zotier et al. 1999, Bianchi y Morri 2000). Hasta hace pocos años, la recolección de huevos y ejemplares juveniles y adultos, por la población local, era una de las amenazas más significativas (James 1984, Mayol 1986). Sin embargo, actualmente la disponibilidad de alimento y de zonas de cría son los factores ecológicos más importantes en la conservación de las aves marinas mediterráneas (Mínguez et al. 2003). La captura incidental en artes de pesca también es un factor potencialmente importante, y hasta hace poco ignorado, que podría afectar gravemente a algunas poblaciones y especies reduciendo sus densidades a niveles preocupantes (Valeiras y Camiñas 2003, Cooper et al. 2003). La gestión de las actividades pesqueras y la protección del hábitat de nidificación son difíciles de gestionar debido a una legislación dispersa, el gran número de administraciones implicadas y los intereses económicos.

La mayor parte de los problemas de conservación están identificados pero las acciones de conservación eficaces son escasas (Mínguez et al. 2003).

(II) La especie: la pardela balear *Puffinus mauretanicus*, el ave marina más amenazada del Mediterráneo

La pardela balear *Puffinus mauretanicus* (Lowe 1921) es el ave marina más amenazada del Mediterráneo, catalogada como En Peligro Crítico según los criterios de la IUCN (Arcos y Oro 2004, BirdLife International 2006) (ver Figura 1.3). Dos factores hacen que su estado de conservación sea delicado: una distribución reducida a las Islas Baleares y una población reproductora limitada (inferior a 2000 parejas) (Krebs 2001, Arcos y Oro 2004, Ruiz y Martí 2004, BirdLife International 2006) (ver Figura 1.3).



Figura 1.3 Pardela balear *Puffinus mauretanicus*

Ecología. La pardela balear, de la familia de los Procellariiformes, es un ave marina de tamaño medio, 500 g aproximadamente, que presenta un ligero dimorfismo sexual (los machos son en promedio un 5% mayores que las hembras, Genovart et al. 2003). Nidifica en grietas, cuevas y agujeros, en colonias relativamente pequeñas, desde una sola pareja aislada a cuevas que suelen albergar varias parejas, a diferentes alturas sobre el nivel del mar (Ruiz y Martí 2004). Las colonias de cría se localizan en tramos de acantilado rocoso de todo el litoral balear, no sólo en islotes sino también en las grandes islas como Formentera, Mallorca o Menorca, aunque existen también citas de nidificación tierra adentro (de hasta 500m de la costa) en Eivissa y Cabrera (Alcover et

al. 1994, Aguilar 2000, Arcos y Oro 2004) (ver Figura 1.2). Las pardelas pueden nidificar sobre sustrato duro o blando (en ocasiones llegando a excavar ligeramente) y aportan pequeñas piedras, huesos u objetos que encuentran en las cercanías. Las cópulas se producen mayoritariamente un mes antes que las puestas, seguido de un éxodo preposital de corta duración (Ruiz y Martí 2004). Las puestas, de un único huevo, se realizan entre finales de febrero y principios de marzo, los pollos eclosionan a los 48-52 días (entre mediados de abril y principios de mayo) y vuelan a los 65-70 días (entre finales de junio y principios de julio).

La pardela balear es una especie de hábitos gregarios que se alimenta de cardúmenes de pequeños peces pelágicos (boquerón *Engraulis encrasicolus* y sardina *Sardina pilchardus* principalmente, aunque también de cefalópodos de migración vertical) sobre la plataforma continental (Mayol 2000, Arcos y Oro 2002). Es una especie buceadora, con zambullidas de hasta 66 segundos, que alcanza un máximo de profundidad conocida de 26 m (Aguilar et al. 2003).

La plataforma Ibérica, y principalmente la zona de influencia del río Ebro, es una de las principales áreas de alimentación de la especie durante el periodo reproductor (Arcos y Oro 2002, Abelló et al. 2003), debido a su alta productividad marina aunque se han señalado otras importantes zonas como la costa africana y la plataforma continental del archipiélago balear (Ruiz y Martí 2004) (Figura 1.2). Un modelo bioenergético estimó que más del 40% de los requerimientos energéticos de la especie, durante el final de la época de cría (mayo-junio), los cubrían los descartes de los arrastreros (Arcos y Oro 2002). De forma oportunista, la pardela balear puede alimentarse de las comunidades de peces asociados a objetos flotantes a la deriva, así como asociarse a depredadores sub-superficiales, como delfines y atunes, ya que éstos conducen cardúmenes de pequeños peces pelágicos hacia la superficie y las pardelas pueden acceder a los mismos (Oro 1995, Arcos et al. 2000, Arcos y Oro 2002). Durante la noche pueden alimentarse de organismos planctónicos y depredadores asociados, relacionados con su migración vertical diurna (Arcos et al. 2000, Arcos y Oro 2002).

Después de la cría, la pardela balear migra hacia otras zonas del Mediterráneo o pasa a través del estrecho de Gibraltar, rodeando la península Ibérica, hasta Galicia y desde aquí se separa de la costa alcanzando el sur de Gran Bretaña y el oeste de Francia, donde realiza la muda (Mayol et al. 2000, Ruiz y Martí 2004). En los últimos años se ha detectado un cambio en los patrones de distribución postnupcial registrándose un mayor número de individuos más al norte (al sur de Inglaterra), probablemente

relacionado con el calentamiento del agua que habría ocasionado cambios en los patrones de distribución de sus presas (Wynn et al. 2005). En las zonas de muda, las aves se alimentan mayoritariamente de pequeños peces pelágicos aunque también aprovechan los descartes pesqueros (Le Mao y Yésou 1989, Mouriño et al. 2003, Arcos y Oro 2004).

Entre finales de agosto y octubre el grueso de la población reproductora vuelve al Mediterráneo occidental (Ruiz y Martí 2004) y la pardela balear se agrega principalmente en aguas del levante Ibérico durante el otoño y el invierno, con una distribución más costera y gregaria que en primavera (Arcos 2001, Arcos 2005). El principal recurso durante este periodo son los pequeños peces pelágicos (capturados generalmente en grupo). La asociación a arrastreros y barcos de cerco es frecuente aunque menor, en términos numéricos, y mayor, respectivamente, si lo comparamos con el periodo reproductor (Arcos y Oro 2002, Arcos 2005). Cabe destacar que individuos inmaduros y no reproductores podrían desplazarse hacia el Atlántico norte a lo largo de todo el año (Ruiz y Martí 2004).

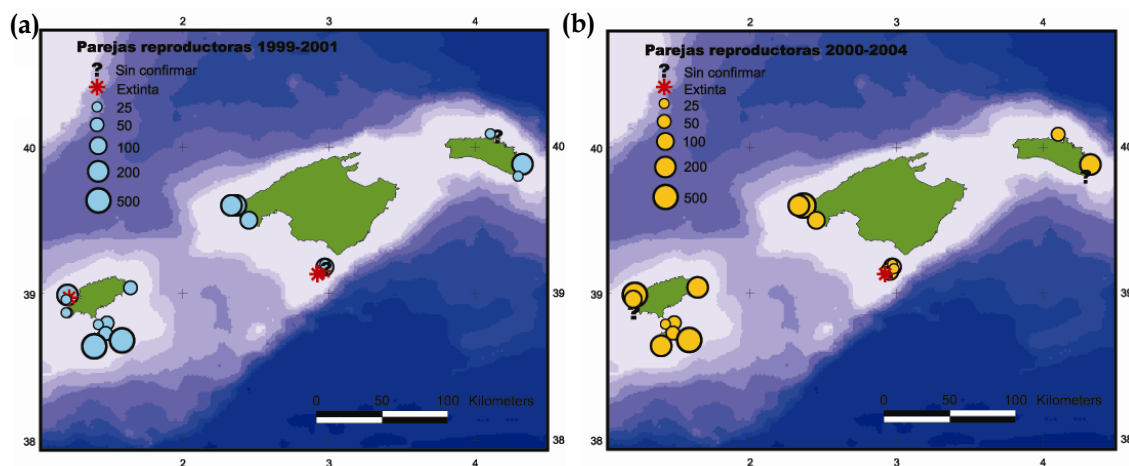


Figura 1.4 Distribución de parejas reproductoras en (a) 1999-2001 y (b) 2000-2004 (adaptado de Ruiz y Martí 2004 y Molina y McMinn 2005, respectivamente).

Amenazas. Para poder realizar una diagnosis de conservación de una especie y valorar el impacto de una amenaza específica primero es necesario obtener estimas poblacionales fiables, o alternativamente poder estimar sus parámetros demográficos (Russell 1999, Caswell 2001). Los hábitos cavernícolas y nocturnos de la pardela balear hacen compleja la estima de la población reproductora total, por el difícil acceso a la mayoría de las colonias y el dificultoso contaje de los nidos ocupados. De hecho, el último dato oficial (1999-2001) estimó la población reproductora mundial en 1750-2125 parejas (23%, 8%, 4%, 13% y 52% en Mallorca, Cabrera, Menorca, Eivissa y Formentera,

respectivamente) (Ruiz y Martí 2004) (ver Figura 1.4a). Estimaciones más recientes (2000-2004, Conselleria de Medi Ambient del Govern Balear) mantienen la población reproductora en ca. 2000 parejas pero han arrojado un cambio en las proporciones por islas (30%, 10%, 10%, 20% y 30% en Mallorca, Cabrera, Menorca, Eivissa y Formentera, respectivamente), incluyendo nuevas localidades de cría (Molina y McMinn 2005) (ver Figura 1.4b).

Las aves marinas son los únicos depredadores marinos que sufren amenazas tanto en el ecosistema marino como en el terrestre, principalmente de origen antropogénico. Así, en las colonias el consumo humano, el uso y la modificación del hábitat de nidificación y las especies introducidas son las mayores amenazas para la pardela balear, mientras que en el mar la pesca y la contaminación son los aspectos más relevantes en cuanto a su conservación. Al igual que otras especies longevas, la mortalidad adulta es el parámetro demográfico que más afecta a la tasa de crecimiento poblacional. Por tanto, la identificación de los factores que afectan negativamente la supervivencia adulta y el diseño e implementación de medidas correctoras efectivas deben ser actuaciones prioritarias en las estrategias y programas de conservación para garantizar la viabilidad de la pardela balear.

En tierra...

En las colonias de cría, el consumo humano y la pérdida y degradación del hábitat eran las causas principales del fuerte declive y extinción local de algunas poblaciones de pardela balear hasta hace pocas décadas (Mayol 1986, Aguilar 1998, Mayol et al. 2000). La distribución actual de las colonias de pardela balear está probablemente condicionada por la presencia de especies depredadoras (Molina y McMinn 2005). Las ratas afectan a la productividad de la especie, al igual que ocurre en otras islas mediterráneas (Martin et al. 2000, Palmer y Pons 2001), mientras que los gatos asilvestrados y mustélidos pueden tener un mayor impacto al depredar directamente sobre adultos (existe un caso de depredación de 21 ejemplares no juveniles de pardela balear en un mes en una colonia de 150-200 parejas reproductoras; Ruiz y Martí 2004).

La ocupación humana y las actividades recreativas, debido al aumento de la población residente y visitante, también son una amenaza importante para la especie (Mayol et al. 2000). Por ejemplo, existen evidencias de que el exceso de luz del alumbrado en las costas cercanas a las colonias de cría reduce las vocalizaciones de las aves, probablemente para evitar atraer a los depredadores (Ruiz y Martí 2004).

Recientemente, se detectaron diferencias en la coloración, morfometría y otras características biológicas en los ejemplares capturados en la mayor colonia de Menorca, en comparación con el resto de las islas (Ruiz y Martí 2004). Estos indicios sugirieron la posibilidad de la presencia de la pardela mediterránea *Puffinus yelkouan* en Menorca, la colonia más septentrional de pardela balear. Recientemente, Genovart et al. (2005, en prensa) constataron dichas diferencias morfométricas, entre ejemplares de esta colonia de Menorca y el resto de las colonias del archipiélago, y estudios genéticos revelaron la existencia de haplotipos tipo *mauretanicus* y *yelkouan* en la colonia de Menorca. Sin embargo, no se encontraron evidencias genéticas de un declive poblacional (endogamia o pérdida de diversidad genética). Otros resultados reveladores indicaron que, a pesar de la fuerte filopatría descrita para los Procellariiformes (Warham 1990), la estructura poblacional era débil, sugiriendo una dispersión natal importante y que el gran desequilibrio encontrado en las tasas de inmigración y emigración sugería una heterogeneidad espacial en la calidad de las colonias. En cuanto a la relación filogenética entre ambas especies, los autores sugieren que la pardela balear divergió de la mediterránea hace 1 millón de años y experimentó una expansión demográfica intensa durante el Pleistoceno medio. Un contacto secundario con la pardela mediterránea habría producido una hibridación interespecífica en la periferia de la distribución de la pardela balear.

En el mar...

En el mar, la pesca es una de las mayores amenazas, ya que puede provocar una alta mortalidad, principalmente en el palangre (Belda y Sánchez 2001, Valeiras y Camiñas 2003) debido al comportamiento gregario de la especie (Arcos y Oro 2004). Se debe también evaluar el impacto de otras artes de pesca al existir evidencias de la captura de un ejemplar juvenil en un arte fijo tradicional calado en la cercanía de una de las colonias de cría, junto a 5 ejemplares juveniles de cormorán moñudo *Phalacrocorax aristotelis*, en el Parque Nacional del Archipiélago de Cabrera el 4 de julio de 2005 (Com. Pers. M McMinn).

La reducción de sus principales presas y la degradación del ecosistema marino, principalmente debida a la pesca de arrastre, son otras de las amenazas importantes para la especie (ver efectos sobre las aves marinas en apartados anteriores) (Arcos y Oro 2004, Arcos et al. 2006). La pardela balear explota de forma importante los descartes de pesca durante la época reproductora, probablemente relacionado con una

fuerte disminución de algunas especies de pequeños peces pelágicos como el boquerón (Arcos y Oro 2004). Sin embargo, el aumento de otras especies como la alacha *Sardinella aurita* (Abad et al. 1998) podría compensar dicha disminución (Arcos y Oro 2004), aunque en los últimos años el declive parece afectar a la mayoría de especies de pequeños pelágicos con una reducción generalizada de su biomasa (Arcos et al. 2006). Una de las estrategias de gestión pesquera prioritarias en la Unión Europea es la reducción de los descartes de pesca aumentando la selectividad de las artes (COM 2002). Las regulaciones pesqueras deberían integrar a las aves marinas, incluyendo la pardela balear, para paliar tanto el potencial impacto de la sobreexplotación de sus presas naturales como el de la reducción de los descartes (Arcos y Oro 2004).

No hay que olvidarse de otras amenazas como la contaminación por metales pesados y organoclorados (Ruiz y Martí 2004). Análisis de plumas de adultos han revelado contaminación por plomo y niveles especialmente altos de selenio y mercurio. En el delta del Ebro, una de las áreas más importantes de alimentación durante la reproducción, los niveles de mercurio en los descartes de pesca son más elevados que en sus presas naturales (Arcos et al. 2002, Arcos y Oro 2004). Los niveles de selenio podrían atenuar el efecto del mercurio aunque también afectar negativamente la reproducción (Arcos y Oro 2004). Los niveles de organoclorados no eran críticos, pero indicaban que la especie estaba afectada por compuestos orgánicos de origen industrial (Ruiz y Martí 2004). En cuanto a la contaminación por hidrocarburos, no se conoce ningún caso de elevada incidencia, pero el alto gregarismo de la especie la hace muy vulnerable a vertidos accidentales. Por ejemplo, si el hundimiento del Prestige se hubiese producido cuando el grueso de la población se encontraba en aguas gallegas (verano) la mortalidad hubiera sido alarmante (Ruiz y Martí 2004).

Estrategias de conservación. El Govern de les Illes Balears elaboró en 1997 el primer Plan de Recuperación para la especie (Aguilar 1998) cuya ejecución la realizó SEO/Birdlife, a través del proyecto *Life B-4/3200/97/246*. Se constató que la viabilidad de la población dependía de la productividad pero principalmente de la supervivencia adulta y que la disponibilidad del hábitat de nidificación no era un factor limitante. Se actualizó la estima de la población reproductora y se avanzó en el conocimiento de los principales parámetros demográficos. Así, entre otros, se procedió a mejorar la calidad del hábitat de reproducción controlando los depredadores, reduciendo las molestias y protegiendo legalmente todas las colonias de cría declarándolas como Zonas de Especial Protección para las Aves (ZEPA).

Después del proyecto, se identificaron futuras actuaciones prioritarias (Ruiz y Martí 2004): (1) control de depredadores introducidos, (2) valoración de la contaminación de selenio y mercurio durante el ciclo anual, (3) control de expolios y molestias, (4) estimar y disminuir la mortalidad en el palangre, (5) cumplir con los planes de gestión de las ZEPAs, (6) analizar la influencia de la sobreexplotación de los recursos pesqueros y (7) declarar ZEPAs en el medio marino.

Cuando una especie se asigna a la categoría de En Peligro Crítico en el Catálogo Nacional, la comunidad/es autónoma/s correspondiente está obligada a elaborar y aprobar un Plan de Recuperación (Ley 4/1989). Por tanto, a partir del primer Plan y los resultados del proyecto *Life*, el Govern de les Illes Balears presentó en 2003 el segundo Plan de Recuperación (2003-2008) de la pardela balear. El objetivo general es invertir la tendencia demográfica de la especie, frenando su regresión actual y favoreciendo el incremento de sus efectivos, disminuyendo los factores que provocan la mortalidad adulta, incrementar la productividad de las colonias, mantener las colonias existentes y recuperar las preexistentes, y mejorar el conocimiento actual de la biología de la especie. También es prioritario sensibilizar a la sociedad del interés de la especie, principalmente en Formentera, y promover la participación de todos los sectores implicados en la recuperación de la especie.

El esfuerzo realizado por las Administraciones y entidades implicadas, tanto en la elaboración de los Planes de Recuperación como en su ejecución, es muy loable. Sin embargo, un estudio demográfico detallado es crucial para estudiar la viabilidad e implicaciones de conservación para esta especie amenazada endémica de las Islas Baleares, al igual que el estudio de su ecología en el ecosistema marino, analizando su asociación al hábitat de alimentación y la interacción con las actividades antropogénicas que se desarrollan en el Mediterráneo occidental.

1.6 Enfoque de la tesis

El principal objetivo de esta tesis doctoral ha sido la de estudiar la problemática de conservación de la pardela balear en el Mediterráneo occidental a través de sus rasgos de historia de vida y sus interacciones con el ecosistema marino. Muchos trabajos de investigación en aves marinas se centran únicamente en alguno de los componentes del hábitat, pero esta tesis doctoral ha pretendido unir puentes sobre el conocimiento de la biología de la especie en el ecosistema terrestre y marino. Esta investigación se ha desarrollado principalmente asociada al proyecto europeo DISCBIRD cuyo objetivo primordial ha sido la de proporcionar información detallada del impacto que los cambios en las tasas de descarte, en aplicación de las políticas pesqueras actuales, pueden tener sobre las comunidades de aves marinas, principalmente en el mar del Norte y Mediterráneo occidental, para poder minimizar los impactos adversos de estas regulaciones.

El trabajo de campo se realizó en las colonias de cría más accesibles del archipiélago balear, principalmente entre 1997-2004 (durante el periodo 1997-2000 el seguimiento se realizó dentro del primer Plan de Recuperación y el proyecto *Life*). Dado el delicado estatus de conservación de la especie, el seguimiento se ha realizado con una enorme cautela, intentando minimizar los efectos derivados. En las áreas de alimentación, el muestreo se realizó tanto en la plataforma continental Ibérica (1999-2002, realizado por JM Arcos) como alrededor de las colonias de cría (archipiélago balear, 2002-2003).

Esta tesis doctoral está dividida en dos secciones en relación a los dos ecosistemas diferentes que ocupa la pardela balear. La primera sección se centra en el trabajo de campo realizado en las colonias de cría que nos ha permitido realizar un estudio demográfico detallado, mientras que la segunda sección versa sobre la caracterización de las zonas de alimentación de la especie y su interacción con las pesquerías en el ecosistema marino.

La primera sección se centra principalmente en realizar, por primera vez, una diagnosis de conservación para conocer la dinámica de población de la especie y conocer los parámetros demográficos que están afectando negativamente a la tasa de crecimiento poblacional de la pardela balear. Así, se ha estimado la supervivencia adulta, rasgo fundamental para la conservación de una especie de larga vida, y se realizó un análisis de viabilidad de la población estimando las probabilidades de extinción y tiempo de vida medio (capítulo 2). El éxito reproductor también puede ser

otro rasgo de historia de vida fundamental para la conservación de la especie, por lo que se ha estimado la contribución de cada uno de estos dos rasgos de historia de vida a la viabilidad de la especie, así como la influencia de las condiciones ambientales (recursos tróficos como los pequeños peces pelágicos y los descartes de pesca) sobre la productividad (capítulo 3). El estudio de la demografía nos permite realizar predicciones sobre la vulnerabilidad y formular estrategias de conservación efectivas para la conservación de la especie.

La comparación de rasgos de historia de vida entre poblaciones locales de la misma especie puede darnos una idea de las adaptaciones a las condiciones ecológicas específicas en cada una de ellas. Así mismo, la variabilidad de los rasgos de historia de vida de poblaciones cercanas podría ser un mecanismo evolutivo para evitar la sincronización de las fluctuaciones poblacionales, importante para impedir la extinción simultánea de poblaciones locales y con consecuencias fatales para su conservación. Para ello, se realiza un análisis espacio-temporal de parámetros reproductores como el éxito de eclosión y reproductor (capítulo 3) y tamaño de huevo (capítulo 4) entre diferentes poblaciones locales de pardela balear. En el caso del tamaño del huevo, se analizan factores potenciales que podrían influir en su variabilidad, como el tamaño corporal y la ecología trófica.

La segunda sección se centra en estudiar la asociación de la pardela balear al hábitat de alimentación en el mar y su interacción con la pesca. El estudio de la ecología de la especie en las zonas de alimentación es imprescindible para conocer con detalle las amenazas a las que se enfrenta, sobre todo en un contexto altamente modificado por el ser humano como es el Mediterráneo occidental, durante un periodo del ciclo reproductor crítico como es el periodo de crecimiento de los pollos (capítulo 5). La identificación, a través de modelos, de las variables que se correlacionan con el hábitat nos puede ayudar a predecir la distribución de esta especie críticamente amenazada y la magnitud de determinados impactos. Así, se utiliza la asociación al hábitat de la especie para delimitar el área de campeo e identificar áreas de alimentación importantes, caracterizados por altas densidades, y predecir aquellas áreas marinas importantes para la conservación de la especie.

Por último, se sabe que la pardela balear se aprovecha de los descartes de pesca en zonas distantes a las colonias de cría, como el delta del Ebro, aunque no se conoce con exactitud el grado de aprovechamiento de este recurso en el archipiélago balear. Por tanto, se estudia la explotación de este recurso por parte de la comunidad de aves

marinas del archipiélago balear, haciendo especial hincapié en la especie de estudio (capítulo 6). El trabajo de campo se realizó a lo largo de todo el año ya que durante algunos periodos, especialmente fuera de la época de cría, la disponibilidad de recursos tróficos puede ser importante para la supervivencia. Se caracterizan los patrones de distribución y estructura de la comunidad de aves marinas desde una aproximación global, en relación a variables relacionadas con la pesquería, y otra más específica que permite explorar patrones espacio-temporales. Finalmente, se realizan recomendaciones para integrar la interacción de las aves marinas con la pesca en el manejo de las pesquerías del Mediterráneo occidental, ya que estas últimas no sólo afectan a los stocks de peces sino también al ecosistema marino en su conjunto.

Referencias

- Abad R, Miquel J, Iglesias M, Alvarez F (1998) Acoustic estimation of abundance and distribution of anchovy in the NW Mediterranean. *Scientia Marina* 62:37-43
- Abelló P, Arcos JM, Gil de Sola L (2003) Geographical patterns of seabird attendance to a research trawler along the Iberian Mediterranean coast. *Scientia Marina* 67S2:69-75
- Agostini VN, Bakun A (2002) 'Ocean triads' in the Mediterranean Sea: physical mechanisms potentially structuring reproductive habitat suitability (with example application to European anchovy, *Engraulis encrasicolus*). *Fisheries Oceanography* 11:129-142
- Aguilar JS (1998) Biología y conservación de la Pardela Balear, *Puffinus mauretanicus*. Documents Tècnics de Conservació (II época) nº 2, 1997. Conselleria de Medi Ambient, Ordenació de Territori i Litoral. Govern Balear. Palma
- Aguilar JS (2000) La población de pardela balear *Puffinus mauretanicus* en el Parque Nacional del Archipiélago de Cabrera. En: Pons G (Ed) Las Aves del Parque Nacional Marítimo-terrestre del Archipiélago de Cabrera (Islas Baleares, España). Ministerio de Medio Ambiente, Palma de Mallorca, pp. 33-44
- Aguilar JS, Benvenuti S, Dall'Antonia L, McMinn-Grivé M, Mayol-Serra J (2003) Preliminary results on the foraging ecology of Balearic shearwaters (*Puffinus mauretanicus*) from bird-borne data loggers. *Scientia Marina* 67S2:129-134
- Alcover JA, McMinn M, Altaba CR (1994) Eivissa: a Pleistocene oceanic-like island in the Mediterranean. *National Geographic Research & Exploration* 10:236-248

-
- Allen JI, Somerfield PJ, Siddorn J (2002) Primary and bacterial production in the Mediterranean Sea: a modeling study. *Journal of Marine Systems* 33-34:473-495
- Anderson DJ, Huyvaert KP, Wood DR, Gillikin CL, Frost BJ, Mouritsen H (2003) At-sea distribution of Waved Albatrosses and the Galápagos Marine Reserve. *Biological Conservation* 110:367-373
- Arcos JM (2001) Foraging ecology of seabirds at sea: significance of commercial fisheries in the NW Mediterranean. Doctoral dissertation. Universitat de Barcelona. Available: http://tdcat.cesca.es/TESIS_UB/AVAILABLE/TDX-0219102-114337/TOL38.pdf (March 2002)
- Arcos JM (2005) Seabird distribution along the Iberian Mediterranean coast in Autumn/Winter: preliminary results from research cruises *ECOMED 2003* and *ECOMED 2004*. Instituto Español de Oceanografía (internal report)
- Arcos JM, Oro D (2002) Significance of fisheries discards for a threatened Mediterranean seabird, the Balearic shearwater *Puffinus mauretanicus*. *Marine Ecology Progress Series* 239:209-220
- Arcos JM, Oro D (2004) Pardela Balear, *Puffinus mauretanicus*. In: Madroño A, González C, Atienza JC (Eds) Libro Rojo de las Aves de España. Dirección General para la Biodiversidad- SEO/BirdLife, Madrid, pp. 46-50
- Arcos JM, Massutí E, Abelló P, Oro D (2000) Fish associated with floating drifting objects as a feeding resource for Balearic Shearwaters *Puffinus mauretanicus* during the breeding season. *Ornis Fennica* 77:177-182
- Arcos JM, Ruiz X, Bearhop S, Furness RW (2002) Mercury levels in seabirds and their fish prey at the Ebro Delta (NW Mediterranean): the role of trawler discards as a source of contamination. *Marine Ecology Progress Series* 232: 281-290
- Arcos JM, Louzao M, Oro D (2006) Fishery ecosystem impacts and management in the Mediterranean: seabirds point of view. In: Nielsen J, Dodson J, Friedland K, Hamon T, Hughes N, Musick J, Verspoor E (Eds) Proceedings of the Fourth World Fisheries Congress: Reconciling Fisheries with Conservation. American Fisheries Society, Symposium 49, Bethesda, Maryland, in press.
- Barbraud C, Weimerskirch H, Robertson GG, Jouventin P (1999) Size related life-history traits: insights from a study of snow petrels (*Pagodroma nivea*). *Journal of Animal Ecology* 68:1179-1192

-
- Belda EJ, Sánchez A (2001) Seabird mortality on longline fisheries in the western Mediterranean: factors affecting bycatch and proposed mitigating measures. *Biological Conservation* 98:357-363
- Bianchi CN, Morri C (2000) Marine biodiversity of the Mediterranean Sea: situation, problems and prospects for future research. *Marine Pollution Bulletin* 40:367-376
- BirdLife International (2006) Species factsheet: *Puffinus mauretanicus*. Downloaded from <http://www.birdlife.org> on 14/7/2006.
- Boersma PD, Parrish JK (1998) Threats to seabirds: research, education, and societal approaches to conservation. In: Marzluff JM, Sallabanks R (Eds) *Avian Conservation: Research and Management* Island Press, Washington, DC, pp. 237-259
- Boersma PD, Clark JA, Hillgarth N (2002) Seabird conservation. In: Schreiber EA, Burger J (Eds) *Biology of Marine Birds*. CRC Press, Boca Raton, pp. 559-579
- Botsford LW, Castilla JC, Peterson CH (1997) The management of fisheries and marine ecosystems. *Science* 277:509-515
- Brooks TM, Mittermeier RA, da Fonseca GAB, Gerlach J, Hoffmann M, Lamoreux JF, Mittermeier CG, Pilgrim JD, Rodrigues ASL (2006) Global biodiversity conservation priorities. *Science* 313:5861
- Brothers N, Gales R, Reid T (1999) The influence of environmental variables and mitigation measures on seabird catch rates in the Japanese tuna longline fishery within the Australian Fishing Zone, 1991-1995. *Biological Conservation* 88:85-101
- Burger J, Gochfeld M (1993) Tourism and short-term behavioural responses of nesting Masked, Red-footed, and Blue-footed Boobies in the Galapagos. *Environmental Conservation* 20:255-259
- Burger J, Gochfeld M (1994) Predation and effects on humans on island-nesting birds. In: Nettleship DN, Burger J, Gochfeld M (Eds) *Seabirds on islands: Threats, Case Studies and Action Plans*. International Council on Bird Preservation, Cambridge, England
- Burger J, Gochfeld M (2002) Effects of chemicals and pollution on seabirds. In: Schreiber EA, Burger J (Eds) *Biology of Marine Birds*. CRC Press, Boca Raton, pp. 486-525

- Carscadden JE, Frank KT, Leggett WC (2001) Ecosystem changes and the effects on capelin (*Mallotus villosus*), a major forage species. *Canadian Journal of Fisheries and Aquatic Sciences* 58:73-85
- Caswell H (2001) *Matrix population models*, 2nd edn. Sinauer Press, Sunderland
- Coll M, Palomera I, Tudela S, Sardà F (2006) Trophic flows, ecosystem structure and fishing impacts in the South Catalan Sea, Northwestern Mediterranean. *Journal of Marine Systems* 59:63-96
- COM (2002) 656. Commission of the European Communities. Communication from the Commission to the Council and the European Parliament on the Community Action Plan to reduce discards of fish. Brussels, Belgium
- Cooper JN, Baccetti N, Belda EJ, Borg JJ, Oro D, Papaconstantinou C, Sánchez A (2003) Seabird mortality from longline fishing in the Mediterranean Sea and Macaronesian waters: a review and a way forward. *Scientia Marina* 67S2:57-64
- Croxall JP, Rothery P (1991) Population regulation of seabirds: implications of their demography for conservation. In: Perrins CM, Lebreton J-D, Hiron GJM (Eds) *Bird Population Studies, Relevance to Conservation and Management*. Oxford University Press, Oxford, pp. 272-296
- Czech B, Krausman PR, Devers PK (2000) Economic associations among causes of species endangerment in the United States. *BioScience* 50:593-601
- Dhondt AA (2001) Tradeoffs between reproduction and survival in tits. *Ardea* 89:155-166
- Estrada M (1996) Primary production in the northwestern Mediterranean. *Scientia Marina* 60:55-64
- Estrada M, Vives F, Alcaraz M (1985) Life and the productivity of the open ocean. In: Margalef R (Ed) *Western Mediterranean*. Pergamon Press, Oxford, pp. 148-197
- Farrugio H, Oliver P, Biagi F (1993) An overview of the history, knowledge, recent and future trends in Mediterranean fisheries. *Scientia Marina* 57:105-119
- Fazey I, Fischer J, Lindenmayer DB (2005) Who does all the research in conservation biology? *Biodiversity and Conservation* 14:917-934
- Furness RW (1999) Are industrial fisheries a threat to seabird populations? In: Adams NJ, Slotow RH (Eds) *Proceedings 22nd International Ornithological Congress, Durban*. Birdlife South Africa, Johannesburg, pp. 676-687
- Furness RW (2003) Impacts of fisheries on seabird communities. *Scientia Marina* 67S2:33-45

-
- Furness RW, Greenwood JJD (1993) Birds as monitors of environmental changes. Champan and Hall, London
- Furness RW, Camphuysen CJ (1997) Seabirds as monitors of the marine environment. ICES Journal of Marine Science 54:726-737
- Garthe S, Camphuysen CJ, Furness RW (1996) Amounts of discards by commercial fisheries and their significance as food for seabirds in the North Sea. Marine Ecology Progress Series 136:1-11
- Genovart M, McMinn M, Bowler D (2003) A discriminant function for predicting sex in the endangered Balearic shearwater (*Puffinus mauretanicus*). Waterbirds 26:72-76
- Genovart M, Juste J, Oro D (2005) Two sibling species sympatrically breeding: a new conservation concern for the critically endangered Balearic shearwater. Conservation Genetics 6: 601-606.
- Genovart M, Oro D, Juste J, Bertorelle G (en prensa) What genetics tell us about the conservation of the critically endangered Balearic Shearwater? Biological Conservation
- Gerber LR, Hyrenbach KD, Zacharias MA (2005) Do the largest reserves protect whales or whalers? Science 307:525-526
- Gilman E (2001) Integrated management to address the incidental mortality of seabirds in longline fisheries. Aquatic Conservation 11:391-414
- Hanski I (1998) Metapopulation dynamics. Nature 396:41-49
- Harrison S (1994) Metapopulations and conservation. In: Edwards PJ, May RM, Webb NR (Eds) Large-scale ecology and conservation biology. Blackwell Scientific Press, Oxford, pp. 111-128
- Hendriks IE, Duarte CM, Carlo HR (2006) Biodiversity research still grounded. Science 312:1715
- Heppell SS, Heppell SA, Read AJ, Crowder LB (2005) Effects of fishing long-lived marine organisms. In: Norse EA, Crowder LB (Eds) Marine Conservation Biology. Island Press, Washington, USA, pp. 211-231
- Heubeck M, Camphuysen KCJ, Bao R, Humple D, Sandoval A, Cadiou B, Bräger S, Thomas T (2003) Assessing the impact of major oil spills on seabird populations. Marine Pollution Bulletin 46:900-902
- Hutchings JA (2000) Collapse and recovery of marine fishes. Nature 406:882-885
- Hyrenbach KD, Forney KA, Dayton PK (2000) Marine protected areas and ocean basin management. Aquatic Conservation 10:437-458

- Hyrenbach KD, Keiper C, Allen SG, Ainley DG, Anderson DJ (2006) Use of marine sanctuaries by far-ranging predators: commuting flights to the California Current System by breeding Hawaiian albatrosses. *Fisheries Oceanography* 15:95-103
- James PC (1984) The status and conservation of seabirds in the Mediterranean Sea. In: Croxall JP, Evans PGH, Schreiber RW (Eds) *Status & Conservation of the world's seabirds*. International Council for Bird Preservation, pp. 371-375
- James MC, Ottensmeyer CA, Myers RA (2005) Identification of high-use habitat and threats to leatherback sea turtles in northern waters: new directions for conservation. *Ecology Letters* 8:195-201
- Jackson JBC, Kirby MX, Berger WH, Bjorndal KA, Botsford LW, Bourque BJ, Bradbury RH, Cooke R, Erlandson J, Estes JA, Hughes TP, Kidwell S, Lange CB, Lenihan HS, Pandolfi JM, Peterson CH, Steneck RS, Tegner MJ, Warner RR (2001) Historical overfishing and the recent collapse of coastal ecosystems. *Science* 293:629-638
- Jenkins M, Green RE, Madden J (2003) The challenge of measuring global change in wild nature: are things getting better or worse? *Conservation Biology* 17:20-23.
- Johannes RE, Hatcher BG (1986) Shallow tropical marine environments. In: Soulé ME (Ed) *Conservation Biology: The Science of Scarcity and Diversity*. Sinauer Associates, Sunderland, Massachusetts, USA, pp. 371-382
- Krebs CJ (2001) *Ecology: the experimental analysis of distribution and abundance*. Benjamin Cummings, San Francisco
- Le Mao P, Yésou P (1989) The annual cycle of Balearic shearwaters and western Mediterranean yellow-legged gulls: some ecological considerations. In: Aguilar JS, Monbailliu X, Paterson AM (Eds) *Estatus y conservación de aves marinas*. SEO/Birdlife y MEDMARAVIS, Madrid, pp.135-145
- Leslie HM (2005) A synthesis of marine conservation planning approaches. *Conservation Biology* 19:1701-1713
- Levin PS, Kochin BF (2004) Publication of marine conservation papers: is conservation biology too dry? *Conservation Biology* 18:1160-1162
- Lewison RL, Crowder LB, Read AJ, Freeman SA (2004) Understanding impacts of fisheries bycatch on marine megafauna. *Trends in Ecology and Evolution* 19:598-604

- Lipcius RN, Crowder LB, Morgan LE (2005) Metapopulation structure and marine reserves. In: Norse EA, Crowder LB (Eds) *Marine Conservation Biology*. Island Press, Washington, USA, pp. 328-345
- Lleonart J, Maynou F (2003) Fish stock assessment in the Mediterranean: state of the art. *Scientia Marina* 67S1:37-49
- Lloret J, Lleonart J, Solé I, Fromentin J-M (2001) Fluctuations of landings and environmental conditions in the north-western Mediterranean Sea. *Fisheries Oceanography* 10:33-50
- Loope LL, Mueller-Dombois D (1989) Characteristics of invaded islands, with special reference to Hawaii. *SCOPE* 37:257-280
- Malcolm JR, Liu CR, Neilson RP, Hansen L, Hannah L (2006) Global warming and extinctions of endemic species from biodiversity hotspots. *Conservation Biology* 20:538-548
- Martin JL, Thibault J-C, Bretagnolle V (2000) Black rats, island characteristics, and colonial nesting birds in the Mediterranean: consequences of an ancient introduction. *Conservation Biology* 14:1452-1466
- Martínez-Abraín A, Velando A, Genovart M, Gerique C, Bartolomé MA, Villuendas E, Sarzo B, Oro D (2006) Sex-specific mortality of European shags *Phalacrocorax aristotelis* during the Prestige oil spill: demographic implications for the recovery of colonies. *Marine Ecology Progress Series* 318:271-276
- Mayol J (1986) Human impact on seabirds in the Balearic Islands. In: Medmaravis, Monbailou X (Eds) *Mediterranean Marine Avifauna: Population Studies and Conservation*. Springer-Verlag, Berlin, pp. 379-396
- Mayol J, Aguilar JS, Yésou P (2000) The Balearic Shearwater *Puffinus mauretanicus*: status and threats. In: Yésou P, Sultana J (Eds) *Monitoring and Conservation of Birds, Mammals and Sea Turtles of the Mediterranean and Black Seas*. Environment Protection Department, Malta, pp. 24-37
- Meffe GK, Carroll RC (1994) *Principles of Conservation Biology*, 2nd ed. Sinauer Associates, Sunderland, MA
- Millot C (1999) Circulation in the western Mediterranean Sea. *Journal of Marine Systems* 20:423-442
- Mínguez E, Oro D, de Juana E, Martínez-Abraín A (2003) Mediterranean seabird conservation: what can we do? *Scientia Marina* 67S2:3-6

- Molina AR, McMinn MG (2005) Population and distribution of the breeding colonies of the Balearic shearwater *Puffinus mauretanicus*. Second International Manx Shearwater Workshop, Belfast, Northern Ireland
- Monaghan P (1996) Relevance of the behaviour of seabirds to the conservation of marine environments. *Oikos* 77:227-237
- Montevecchi WA (2002) Interactions between fisheries and seabirds. In: Schreiber EA, Burger J (Eds) *Biology of Marine Birds*. CRC Press, Boca Raton, pp. 527-557
- Mouriño J, Arcos F, Salvadores R, Sandoval A, Vidal C (2003) Status of the Balearic shearwater (*Puffinus mauretanicus*) on the Galician coast (NW Iberian Peninsula). *Scientia Marina* 67S2:135-142
- Musick JA (1999) Ecology and conservation of long-lived marine animals. *American Fisheries Society Symposium* 23:1-10
- Myers RA, Worm B (2003) Rapid worldwide depletion of predatory fish communities. *Nature* 423:280-283
- Nel DC, Nel JL, Ryan PG, Klages NTW, Wilson RP, Robertson G (2000) Foraging ecology of grey-headed mollymawks at Marion Island, southern Indian Ocean, in relation to longline fishing activity. *Biological Conservation* 96:219-231
- Nisbet ICT (1994) Effects on pollution on marine birds. In: Nettleship DN, Burger J, Gochfeld M (Eds) *Seabirds on Islands: Threats, Case Studies and Action Plans*. International Council on Bird Preservation, Cambridge, England
- Norris K (2004) Managing threatened species: the ecological toolbox, evolutionary theory and declining-population paradigm. *Journal of Applied Ecology* 41:413-426
- Norse EA, Crowder LB (Eds) (2005) *Marine Conservation Biology. The science of maintaining the sea's biodiversity*. Island Press, Washington, USA
- Norse EA, Crowder LB, Gjerde K, Hyrenbach KD, Roberts C, Soulé ME (2005) The potential for reserves as an ecosystem-based management tool in the open ocean. In: Norse EA, Crowder LB (Eds) *Marine Conservation Biology*. Island Press, Washington, USA, pp. 302-327
- Olivar MP, Salat J, Palomera I (2001) A comparative study of spatial distribution patterns of the early stages of anchovy and pilchard in the NW Mediterranean Sea. *Marine Ecology Progress Series* 217:111-120
- Orians GH, Soulé ME (2001) Whither conservation biology research? *Conservation Biology* 15:1187-1188

- Oro D (1995) Audouin's Gulls *Larus audouinii* associate with sub-surface predators in the Mediterranean Sea. *Journal für Ornithologie* 136:465-467
- Oro D (1996) The effects of trawler discard availability on the egg-laying and the breeding success of the Lesser Black-backed Gull *Larus fuscus* in western Mediterranean. *Marine Ecology Progress Series* 132:43-46
- Oro D (1999) Trawler discards: a threat or a resource for opportunistic seabirds? In: Adams NJ, Slotow RH (Eds) *Proceedings of the 22nd International Ornithological Congress, Durban*. Birdlife South Africa, Johannesburg, pp. 717-730
- Oro D (2003) Managing seabird metapopulations in the Mediterranean: constraints and challenges. *Scientia Marina* 67S2: 13-22
- Oro D, Furness RW (2002) Influence of food availability and predation on survival of kittiwakes. *Ecology* 83:2516-2528
- Oro D, Bosch M, Ruiz X (1995) Effects of a trawling moratorium on the breeding success of the yellow-legged gull *Larus cachinnans*. *Ibis* 137:347-349
- Oro D, Cam E, Pradel R, Martínez-Abraín A (2004) Influence of food availability on demography and local population dynamics in a long-lived seabird. *Proceedings of the Royal Society of London Series B* 271:387-396
- Palmer M, Pons GX (2001) Predicting rat response on small islands. *Ecography* 24:121-126
- Palomera I (1992) Spawning of anchovy *Engraulis encrasicolus*, in the north-western Mediterranean relative to hydrographic features. *Marine Ecology Progress Series* 79:215-223
- Papaconstantinou C, Farrugio H (2000) Fisheries in the Mediterranean. *Mediterranean Marine Science* 1:5-18
- Pauly D, Christensen V, Guénette S, Pitcher TJ, Sumaila UR, Walters CJ, Watson R, Zeller D (2002) Towards sustainability in world fisheries. *Nature* 418:689-695
- Pauly D, Watson R, Alder J (2005) Global trends in world fisheries: impacts on marine ecosystems and food security. *Philosophical Transactions of the Royal Society of London, Series B* 360:5-12
- Peterson CH, Rice SD, Short JW, Esler D, Bodkin JL, Ballachey BE, Irons DB (2003) Long-term ecosystem response to the Exxon Valdez oil spill. *Science* 302:2082-2086
- Pikitch EK, Santora C, Babcock EA, Bakun A, Bonfil R, Conover DO, Dayton P, Doukakis P, Fluharty D, Heneman B, Houde ED, Link J, Livingston PA, Mangel

- M, McAllister MK, Pope J, Sainsbury KJ (2004) Ecosystem-based fishery management. *Science* 305:346-347
- Reeves SA, Furness RW (2002) Net loss-seabirds gain? Implications of fisheries management for seabirds scavenging discards in the northern North Sea. RSPB, Sandy
- Rippeth TP, Simpson JH, Placer RJ, Garcia M (2002) Current oscillations in the diurnal-inertial band on the Catalanian shelf in spring. *Continental Shelf Research* 22:247-265.
- Roberts CM (2005) Marine protected areas and biodiversity conservation. In: Norse EA, Crowder LB (Eds) *Marine Conservation Biology*. Island Press, Washington, USA, pp. 265-279
- Ruiz A, Martí R (Eds) (2004) *La Pardela Balear*. SEO/Birdlife-Conselleria de Medi Ambient del Govern de les Illes Balears, Madrid
- Russell RW (1999) Comparative demography and life history tactics of seabirds: implications for conservation and marine monitoring. In: Musick JA (Ed) *Life in the slow lane: ecology and conservation of long-lived marine animals*. American Fisheries Society, Bethesda, pp. 51-76
- Ryan PG (1987) The effects of ingested plastic on seabirds: correlations between plastic load and body condition. *Environmental Pollution* 46:119-125
- Ryan PG, Connell AD, Gardner BD (1988) Plastic ingestion and PCBs in seabirds: is there a relationship? *Marine Pollution Bulletin* 19:174-176
- Sabatés A (1996) Distribution pattern of larval fish populations in the Northwestern Mediterranean. *Marine Ecology Progress Series* 59:75-82
- Salat J (1996) Review of hydrographic environmental factors that may influence anchovy habitats in the northwestern Mediterranean. *Scientia Marina* 60:21-32
- Salat J, Garcia MA, Cruzado A, Palanques A, Arín L, Gomis D, Guillén J, de León J, Puigdefàbregas J, Sospedra J, Velásquez ZR (2002) Seasonal changes of water mass structure and shelf-slope exchanges at the Ebro Shelf (NW Mediterranean). *Continental Shelf Research* 22:327-348
- Salomone M (2002) Ecological riches threatened as oil-spill history repeats itself. *Nature* 420:347
- Send U, Font J, Krahnemann G, Millot C, Rhein M, Tintoré J (1999) Recent advances in observing the physical oceanography of the western Mediterranean Sea. *Progress in Oceanography* 44: 37-64

- Soulé ME (1985) What is Conservation Biology? *BioScience* 35:727-734
- Stearns S (1992) *The evolution of life histories*. Oxford University Press, Oxford
- Steele JH (1998) Regime shifts in marine ecosystems. *Ecological Applications* 8:S33-S36
- Stenhouse I, Montevecchi WA (1999) Indirect effects of the availability of forage fish and fisheries discards: gull predation on breeding storm-petrels. *Marine Ecology Progress Series* 184:303-307
- Stocker TF, Raible CC (2005) Climate change: water cycle shifts gear. *Nature* 434:830-833
- Tasker ML, Camphuysen CJ, Cooper J, Garthe S, Montevecchi WA, Blaber SJM (2000) The impacts of fishing on marine birds. *ICES Journal of Marine Science* 57:531-547
- Tuck GN, Polacheck T, Croxall JP, Weimerskirch H (2001) Modelling the impact of fishery by-catches on albatross populations. *Journal of Applied Ecology* 38:1182-1196
- Tudela S (2004) Ecosystem effects of fishing in the Mediterranean: an analysis of the major threats of fishing gear and practices to biodiversity and marine habitats. General Fisheries Commission for the Mediterranean (FAO). *Studies and Reviews* 74
- UNEP (2006) *Marine and coastal ecosystems and human wellbeing: A synthesis report based on the findings of the Millennium Ecosystem Assessment*. 76pp (available at <http://www.maweb.org/en/index.aspx>)
- Valeiras J, Camiñas JA (2003) The incidental capture of seabirds by Spanish drifting longline fisheries in the western Mediterranean Sea. *Scientia Marina* 67S2:65-68
- Velarde E, Ezcurra E, Cisneros-Mata MA, Lavin MF (2004) Seabird ecology, El Niño anomalies, and prediction of sardine fisheries in the Gulf of California. *Ecological Applications* 14:607-615
- Vlietstra LS, Parga JA (2002) Long-term changes in the type, but not amount, of ingested plastic particles in short-tailed shearwaters in the southeastern Bering Sea. *Pollution Bulletin* 44:945-955
- Votier SC, Furness RW, Bearhop S, Crane JE, Caldow RWG, Catry P, Ensor K, Hamer KC, Hudson AV, Kalmbach E, Klomp NI, Pfeiffer S, Phillips RA, Prieto I, Thompson DR (2004) Changes in fisheries discard rates and seabird communities. *Nature* 427:727-730

- Warham J (1990) *The Petrels. Their Ecology and Breeding Systems*. Academic Press, London, UK
- Weimerskirch H (2002) Seabird demography and its relationship with the marine environment. In: Schreiber EA, Burger J (Eds) *Biology of Marine Birds*. CRC Press, Boca Raton, pp. 115-135
- Wiese FK, Montevecchi WA, Davoren GK, Huettmann F, Diamond AW, Linke J (2001) Seabirds at risk around offshore oil platforms in the North-west Atlantic. *Marine Pollution Bulletin* 42:1285-1290
- Worm B, Sandow M, Oschlies A, Lotze HK, Myers RA (2005) Global patterns of predator diversity in the open oceans. *Science* 309:1365-1369
- Wynn RB, Yésou P, Josey S (2005) The changing status of the Balearic shearwater *Puffinus mauretanicus* in northwest European waters: relationships to sea surface temperature, phytoplankton concentrations and fish abundance. Second International Manx Shearwater Workshop, Belfast, Northern Ireland
- Zotier R, Bretagnolle V, Thibault J-C (1999) Biogeography of the marine birds of a confined sea, the Mediterranean. *Journal of Biogeography* 26:297-313

Section I

Conservation at the breeding colonies

Chapter 2



Modelling demography and extinction risk in the endangered Balearic shearwater

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2.1 Abstract

Several demographic parameters of Balearic shearwater (*Puffinus mauretanicus*) were estimated and used to model extinction probabilities for the species, which is an endemic seabird confined to a Mediterranean archipelago. Adult survival was analyzed through capture-recapture models at two colonies free of adult predators during 1997-2002. Extinction probabilities for the world breeding population were modelled using Monte Carlo simulations of population trajectories, introducing stochasticity both demographic and environmental. Adult survival estimate (0.780, SE = 0.020, 95% Confidence Intervals: 0.739-0.816) was unusually low for a Procellariiform,

suggesting that sources of mortality other than predators (e.g. fishing gears) also occur. A deterministic model showed a declining trend for the world breeding population ($\lambda = 0.952$), and significantly less than 1. Population viability analysis showed that in the presence of environmental and demographic stochasticities, mean extinction time for the world population was estimated at 40.4 years (SE = 0.2), and mean growth rate showed a 7.4% decrease per year. Demographic stochasticity played a more important role than environmental stochasticity in population growth rate.

2.2 Introduction

The Balearic shearwater (*Puffinus mauretanicus* Lowe; family Procellariidae) is one of the rarest seabirds in the world. Recent studies have shown that birds breeding at the Balearic archipelago (western Mediterranean) are genetically and morphologically distinct from the Levantine shearwaters *P. yelkouan*, and is now considered a different species, rather than a subspecies or a race (Austin 1996, Mayol 1997, Heidrich et al. 1998, Heidrich et al. 2000). This has implied a very small breeding distribution, and together with a dramatic decline in the estimated numbers (world breeding population is estimated at less than 2000 pairs), has lead researchers to consider the species as threatened (Arcos & Oro 2004). Although several studies have recently analyzed several features of the ecology of the species (e.g. Aguilar 2000, Arcos et al. 2000, Arcos & Oro 2002), demographic information is sparse. Some data have been recorded on hatching and breeding success although sample sizes were always small (e.g. Aguilar 1992, 1993, 1997). It is also known that some factors are negatively affecting the population dynamics of Balearic shearwaters and other seabird species breeding at the same region (Mayol 1986, Cooper et al. 2003). Even though human predation and loss of suitable habitat were probably the main causes of heavy decline and local extinction of most of these species up to recent decades (Mayol 1986, Aguilar 1997, Alcover 2000, Mayol et al. 2000), other factors such as predation by alien terrestrial mammals (both rats and carnivores) or mortality at fishing gears have been proposed to be now the main threat (Aguilar 2000, Mayol et al. 2000, Belda & Sánchez 2001). However, very little is known about the impact of these factors on the population dynamics of Balearic shearwaters (Arcos & Oro 2004). Thus, a detailed demographic study is crucial to assess population viability and conservation implications for this rare species (Lebreton & Clobert 1991, Caswell 2001).

In this study, we estimated for the first time adult survival of Balearic shearwaters. Owing that all Procellariiformes are long-lived organisms (e.g. Croxall & Rothery 1991; Russell 1999) adult survival is the demographic parameter suspected to have the greatest impact on population growth rate. We used this estimation and other life history traits to perform a population viability analysis and to assess the risk of extinction and expected lifetime for the total world population.

2.3 Material and Methods

Study area

The study was conducted during 1997-2002 at two colonies located in Mallorca (western Mediterranean). One colony (Sa Cella) was located in a big cave at the foot of a vertical sea cliff on Mallorca Island and was accessible only from the sea. The other colony was spread in several small caves on a calcareous rocky islet (Conills de Malgrats) south of Mallorca. While the first cave is free of carnivores and rats, Black rats (*Rattus rattus*) are present in the second colony. Rats are known to predate on Procellariiformes, especially on eggs and chicks, and even on adults (e.g. Warham 1990). Nevertheless, several eradication campaigns have been carried out at Conills colony, and density of rats is probably low, which in turn may determine low or even absence of predation (Amengual & Aguilar 1998, Aguilar 2000). Moreover, Balearic shearwaters are sufficiently large to be safe against rats, and have not found indications of rat predation on eggs or chicks at Conills colony during the study (43 nests monitored, see below). The two colonies are legally protected and no human impact on the colonies has been recorded in the past decade. Breeding pairs were estimated at ca. 170-210 at the big cave and 40-50 at the islet (own data).

Body mass of Balearic shearwaters is ca. 500g, and males are slightly larger than females (Genovart et al. 2003). Females lay a single egg at a nest site (a burrow, a crevice or directly on bare ground) during February-March, and incubation lasts ca. 50 days. Chick rearing period extends for ca. 65-70 days (own data).

Data collection and demographic analysis

Nests were marked and breeding success (as percentage of fledglings by eggs laid) was estimated at the two colonies during 2001-2002. Only nests where the whole breeding cycle was monitored were considered. Productivity was constant between years and colonies (Table 2.1) (three-dimensional contingency table, $\chi^2 = 3.43$, d.f. = 4, $P > 0.05$).

Table 2.1 Fecundity of Balearic shearwaters at the two study colonies during 2001-2002, expressed as the proportion of nesting attempts (pairs laying an egg) that produced a fledgling (n showed the number of nests monitored).

Year	Sa Cella	n	Conills	n	Total	n
2001	0.62	107	0.48	23	0.59	130
2002	0.59	91	0.75	20	0.62	111
Total	0.61	198	0.60	43	0.61	241

A ringing programme on chicks and adults has been carried out since 1981 at several colonies. In total, 880 adults and 1092 chicks have been caught and ringed. However, the limited number of recaptures in most of these years and colonies precluded a complete analysis of the data. Only data from the two study colonies during 1997-2002, when the number of recaptures was sufficiently large, was consequently used. Birds were trapped by hand at the caves mostly at night when individuals visit the colonies. During 1997-1999 we took into account all the individuals trapped at holes and crevices (i.e. potential nest sites) only during the breeding season (from early March to late June), although some of them could be non-breeding individuals (either immatures or adults, see later). We added data from 2000-2002, but only for those individuals recorded as breeders (incubating an egg or with a chick), since this information was systematically recorded during this period. Birds were marked with monel rings with a unique code and the code was recorded when the birds were caught in subsequent years.

Survival estimates by capture-recapture modelling. Capture-recapture modelling was applied to the data set to estimate survival. Models rely on the assumption that all marked individuals have the same survival probability, regardless of whether they were marked at the sample occasion immediately before the current resighting (e.g., Burnham et al. 1987, reviewed in Williams et al. 2002). Nevertheless, several sources of heterogeneity in survival probability among individuals violate this assumption: one of

these sources is the presence of transients. Since birds caught during the period 1997-1999 could include some non-breeders (see earlier), we took into account the potential bias of this effect for these years. Transients are individuals that leave the study area after first capture and thus have a subsequent local survival probability equal to 0 (Pradel et al. 1997). The presence of transients in the marked sample may bias estimation of survival. Comparisons between models including a residency parameter $(1-\tau) < 1$ (not all the individuals in the sample are residents) or equal to 1 (the marked sample includes residents only) addresses the presence of transients in the study colonies (Hines 1996). Importantly, the presence of transients indicates emigration out of the study area or in our case, occurrence of animals simply visiting the caves, but not able to breed (see earlier). This parameter can have a biological meaning such as dispersal probability (e.g. Oro et al. 1999) but in our study the only matter of interest is birds that are breeders (i.e. the estimation of adult survival). However, as usual, we cannot tell to what extent the local survival probability of individuals resighted several times reflects true survival (i.e., it may also incorporate permanent emigration out of the study area). Thus, local adult survival estimation cannot distinguish mortality from permanent emigration. Nevertheless, we assumed that this bias was small in our species since breeding dispersal is probably a very uncommon phenomenon as recorded for other Procellariiformes (e.g. Warham 1990, Aguilar 2000).

We started with the Cormack-Jolly-Seber (CJS) model $(\phi_{t,c}, p_{t,c})$, where ϕ is the local survival rate and p is the recapture probability (e.g., Pollock et al. 1990, Lebreton et al. 1992). This model had time-specific (t) parameters for each colony (c), and can be considered the general model for departure (see details in Lebreton et al. 1992). For model selection and assessment of the biological hypotheses guiding model development (Burnham & Anderson 1998), it is critical that the general model fit the data (Lebreton et al. 1992). Consequently, we first assessed the fit of the general model using the program U-Care (Choquet et al. 2000). The program provides insight into possible violations of the assumptions underlying the CJS model (see Burnham et al. 1987, Pradel et al. 1997). Two tests are used to assess whether these assumptions are met in our CJS initial model: TEST2 and TEST3. TEST2 deals with capture heterogeneity, and check whether some individuals can have different recapture probabilities than others. TEST3 tests the assumption that all marked individuals alive at (i) have the same probability of surviving to ($i+1$), and test specifically the presence of transients (or non-breeders in our case). Analyses were performed using program

MARK (White & Burnham 1999). Model notation followed Lebreton et al. (1992). A model including two factors and their interaction was described using an asterisk. Additive models were described using a “+” symbol instead, while “·” indicated that the parameter was constant with time. The biological hypotheses underlying the structure of models were assessed using AICc (Akaike Information Criterion corrected by the quasi-likelihood parameter \hat{c} , see White & Burnham 1999). Models with the lowest values of AICc were retained as good candidate models (Lebreton et al. 1992, Anderson & Burnham 1999). Models with differences in AICc values lower than 4 were considered as “equivalent” (i.e., data were insufficient to lead to definitive conclusions concerning the process that gave rise to the data) (Anderson et al. 1994, Burnham et al. 1995). To account for the model selection uncertainty, we estimated model parameters by the model averaging procedure, which considers the relative importance of each fitted model (Burnham & Anderson 1998).

We finally partitioned the variance of the apparent survival rates into sampling variance and process variance using random effects models, since a large sampling variance makes difficult the identification of the biological processes underlying the variability in survival (e.g. Burnham & White 2002).

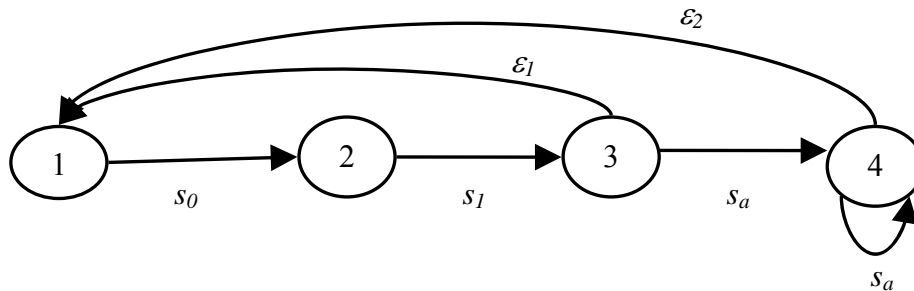


Figure 2.1 Life-cycle representation of the population model. Stage 1 is juvenile birds of age one, stage 2 is immature birds of age two, stage 3 is breeders of age three, and stage 4 is breeders of older ages; s_i are survival for each age class i (juvenile, immature and adult); fertility ε is different for stage 3 (ε_1) than for stage 4 (ε_2) because proportion of breeders (β) was different between the two stages (see Methods for explanations).

Population viability modelling. Population viability analysis and risk of extinction of the world breeding population were assessed by population modelling. As initial value in our models, we took the highest of the available estimates of the world breeding population up-to-date: 2000 breeding females (Arcos & Oro 2004). As a first step, we modelled the life-cycle of the population based on its life-history traits. The life-cycle graph corresponded to a prebreeding census (Figure 2.1; Caswell 2001). Demographic parameters were not sex-dependent. The parameters were s_a , adult survival (or

survival of breeders, from the age of 3 years, see Results); s_0 , juvenile survival; s_1 , survival rates for immatures; a , age at first breeding; and ε , fertility. Fertility comprised β , or proportion of breeders, probability of taking a sabbatical year k (see below) and productivity or breeding success P (see earlier for methods). No information was available on immature survival of Balearic shearwaters. Data on survival other than adult age classes in seabirds are sparse, since immature birds do not normally visit the breeding sites and their survival cannot be easily estimated (e.g. Nichols et al. 1990, Lebreton 2001). Thus, we arbitrarily set a progressive decrease of 5% in survival rate for the immature age classes starting from the adult survival estimate calculated by capture-recapture modelling. This can be considered an optimistic procedure since in seabirds, immature and subadult survival is much lower than adult estimates (e.g. Harris et al. 1994, Spendelov et al. 2002). Moreover, only two age classes were considered immature since birds start to breed at 3y old (see later).

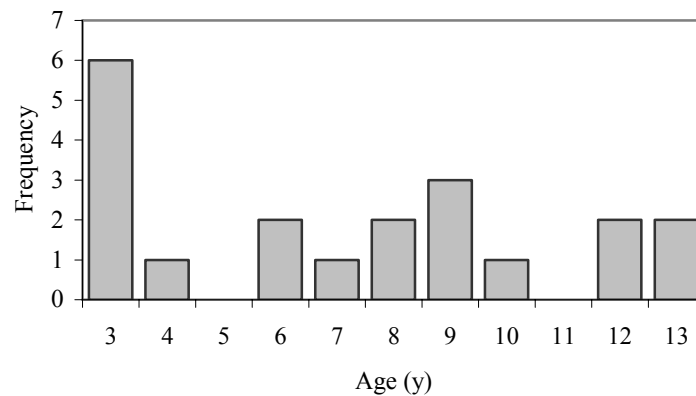


Figure 2.2 Age of first detected breeding of Balearic shearwaters ringed and recaptured at the study colonies. Due to several constraints the bars do not necessarily show the true age of first reproduction (see Methods).

Uncertainty affected also other parameters such as age at first breeding and proportion of breeders. Although ca. 1100 chicks were ringed at several colonies during 1986-2001, very few birds were recovered as breeding adults, and age-specific probabilities of breeding (e.g. Lebreton et al. 1990) could not be estimated. However, 20 of those chicks were first recorded breeding at the study colonies at a mean age of 7.2 years (SD = 3.6), 30% starting at the youngest age of 3 years (Figure 2.2). Distribution was probably biased by several factors (small sample size, detection probabilities, numbers of birds marked per cohort, lack of recaptures for many years), but we could set a minimum age of first breeding at 3 years. We also set an optimistic

approach of 70% proportion of breeders (β) for 3y age class and 100% for older age classes (see Figure 2.1).

Even though many Procellariiformes take sabbatical years (e.g. Warham 1990), the probability of skip breeding in Balearic shearwaters was also uncertain. We obtained a sample of 16 nests where at least data from three consecutive years were recorded. After eliminating data of first and last years when no breeding was recorded (the first case could be a first no-breeding occupation of a nest site and the second case could indicate a definitive abandon of the nest), we estimated the probability that a nest site was unoccupied given that it was occupied in the previous year at 0.213 (SE = 0.060). Assuming no movement among nest sites (own observations), an occupied nest site may become unoccupied in the following year if either a) both birds survived and they skip breeding or b) at least one of the adults died. The probability (k) of an occupied burrow becoming unoccupied could thus be written as:

$$k = p(s_3^2) + (1 - s_3^2), \quad (\text{Eq. 1})$$

where s_3 is adult survival and p the probability of skipping for breeders, which can thus be estimated by:

$$p = \frac{k - 1}{s_3^2} + 1 \quad (\text{Eq. 2})$$

A standard error for this estimate p obtained by the delta method, is

$$SE(p) = (1 - p) \sqrt{\left(\frac{SE(k)}{k - 1}\right)^2 + \left(\frac{2s_3 SE(s_3)}{s_3^2}\right)^2} \quad (\text{Eq. 3})$$

These equations gave estimates of k of 0.261 (SE = 0.063).

Two sets of simulations were carried out: deterministic and stochastic. Deterministic age-structured matrix models were used to test the null hypothesis that the population was stable or increasing, that is $H_0: \lambda \geq 1$, where λ is the population growth rate (Caswell 2001). Estimated population growth rate $\hat{\lambda}$ was calculated from the deterministic matrix. The standard error $SE(\hat{\lambda})$ was calculated from the variances of the matrix entries using the series approximation method (Caswell 2001). The null hypothesis was tested by computing the statistic test z :

$$z = \frac{1 - \hat{\lambda}}{SE(\hat{\lambda})},$$

which under H_0 has a normal distribution with mean 0 and variance 1.

The simple structure of the deterministic model allowed us to assess the relative importance of basic demographic parameters without stretching beyond the limits of the available data. Even though some parameters (such as age at first reproduction) had an unknown degree of uncertainty, recent studies indicate that additional age structure in matrix models has little influence on either population growth rate or perturbation analyses (Heppell et al. 2000, Caswell 2001, Yearsley & Fletcher 2002, Yearsley et al. 2003).

Simulations by stochastic models were used to investigate extinction probabilities. Three different scenarios were considered for modelling extinction probabilities: a) only with demographic stochasticity, b) only with environmental stochasticity, and c) with stochasticity both demographic and environmental. Input parameters of the models were randomly selected from uniform distributions between the upper and lower 95% confidence limits for each demographic parameter. Probabilistic distributions of demographic parameters were applied to add demographic stochasticity, which assigned population sizes to integer values. The same demographic parameters were assigned to males and females, and sex ratio was assumed to be balanced. We calculated the numbers of breeders np using the binomial distribution $B(u, \beta)$, with u the minimum number of males and females and β the proportion of breeders. For environmental stochasticity, each demographic parameter x was obtained from a normal distribution with a mean value and its standard deviation estimated in the field during the study. This stochasticity allowed the parameters to oscillate between bad and good values and environmental effects either negative or positive were thus simulated. We did not consider density dependence in population models (e.g. Yearsley et al. 2003) since population growth rate was negative in all cases (see Results). Finally, to assess the relative importance of both stochasticities we performed two simulations on extinction probabilities, one with only demographic stochasticity and the other with only environmental stochasticity.

From matrix models, we estimated growth-rate sensitivity and elasticity to demographic parameters x_i (Caswell 2001). Monte Carlo simulations were used to draw population trajectories (1000 runs) over time (100 years) and to estimate extinction probabilities, which were computed as the number of extinct trajectories over the total number of trajectories.

Table 2.2 Capture-recapture data for Balearic shearwater ringed and recaptured between 1997 and 2002 at the two study colonies of Mallorca Island: Conills Island and Sa Cella cave.

	Released	Newly marked	Year next recaptured					Never recaptured
			1998	1999	2000	2001	2002	
<i>Conills</i>								
1997	28	28	16	4	2	0	0	6
1998	27	11		16	2	0	0	9
1999	44	24			24	5	2	13
2000	47	19				25	6	16
2001	38	8					21	17
<i>Sa Cella</i>								
1997	69	69	8	7	6	3	3	42
1998	71	63		16	16	10	6	23
1999	52	29			15	17	2	18
2000	56	18				33	7	16
2001	130	67					89	39

2.4 Results

A total of 374 adults (281 at Sa Cella and 93 at Conills) were caught at least once during 1997-2002 (Table 2.2). Program U-Care showed that the CJS model (ϕ_{tc}, p_{tc}) fitted our data (TEST2 + TEST3: $\chi^2 = 24.715$, d.f. = 23, $P = 0.365$). TEST3 showed that there was no transient effect at Sa Cella colony, whereas it could exist such effect at Conills colony (TEST3: $\chi^2 = 3.9217$, d.f. = 4, $P = 0.417$; and $\chi^2 = 8.2951$, d.f. = 4, $P = 0.081$ respectively). We interpreted this result as emigration of new birds to the Conills colony in the first years probably as a consequence of capturing visitors (i.e. non-breeding individuals) among breeders.

The CJS model denoted as (ϕ_{tc}, p_{tc}) (see notation in Lebreton et al. 1992) was used as a starting point for model selection with MARK, although additional models (denoted [$\tau_{c2}, \phi_{\cdot}, p_{\cdot}$]) were constructed to take into account the transient effects on Conills colony (noted as $c2$). The finally selected model (Model 2: [$\phi(\cdot), p(c^*t)$]) indicated that survival was constant and estimated at 0.780 (SE = 0.020, 95% CI: 0.739-0.816) (Table 2.3). Modelling did not show any change in adult survival between the two colonies (Models 4 and 8). Models with a transient effect on Conills colony (e.g. Models 3, 5 or 7) were either retained as good models. Recapture probability estimates under the selected model ranged between 0.158 (SE = 0.052) and 0.881 (SE = 0.058) at Sa Cella colony and 0.669 (SE = 0.082) and 0.769 (SE = 0.093) at Conills colony (Figure 2.3). Effort of recapture (indicated by ef in the models, see Table 2.3), measured as number

of days spent at each colony each year, did not influence recapture probabilities, neither as linear association (Model 12) nor as quadratic effect (Model 13). Random effects models were not well ranked suggesting that sampling variance was small with respect to process variance (Models 14 and 15). The weighted average of adult survival estimate was 0.772 (SE = 0.026, 95% CI: 0.711- 0.823), showing that the estimate given by our finally selected model was robust and reliable.

Table 2.3 Capture-mark-recapture models and their AICc (Akaike Information Criterion corrected by the quasi-likelihood parameter \hat{c}) to estimate adult survival of Balearic shearwaters. Models are ranked according to deviance values. The Δ AICc column corresponds to the difference between the AICc of each model and the AICc of the selected model, which is in bold. ϕ is the apparent survival, p the recapture probabilities, τ the transient probability, t stands for the year effect, c for the colony effect ($c2$ indicates Conills colony) and ef for the recapture effort, measured in days of field work by breeding season; ξ shows the random effects models. The number appearing before model notation is used in the text to refer to different models (see Results).

Model	AICc	Δ AICc	Par	Deviance
1: $\phi(c^*t), p(c^*t)$	1178.932	6.62	18	87.802
2: $\phi(\cdot), p(c^*t)$	1172.317	0.00	11	95.967
3: $\tau(c2^*t), \phi(\cdot), p(c^*t)$	1175.607	3.29	12	95.011
4: $\phi(c), p(c^*t)$	1174.400	2.08	12	95.962
5: $\tau(c2), \phi(\cdot), p(c^*t)$	1174.405	2.09	12	95.966
6: $\phi(t), p(c^*t)$	1177.767	5.45	14	95.129
7: $\tau(c2^*t), \phi(\cdot), p(c^*t)$	1178.006	5.69	16	91.138
8: $\phi(c), p(t)$	1214.235	41.92	7	146.163
9: $\phi(\cdot), p(c)$	1260.147	87.83	3	200.234
10: $\phi(c), p(c)$	1260.166	87.85	4	198.225
11: $\phi(c), p(\cdot)$	1271.611	99.29	3	211.698
12: $\phi(\cdot), p(c^*ef)$	1254.664	82.35	5	190.686
13: $\phi(\cdot), p(c^*ef^2)$	1253.462	81.14	7	185.384
14: $\phi(\cdot), p(c^*t)^\xi$	1909.156	736.84	21.0	893.424
15: $\phi(\cdot), p(c^*ef)^\xi$	1595.122	422.81	15.5	1591.136

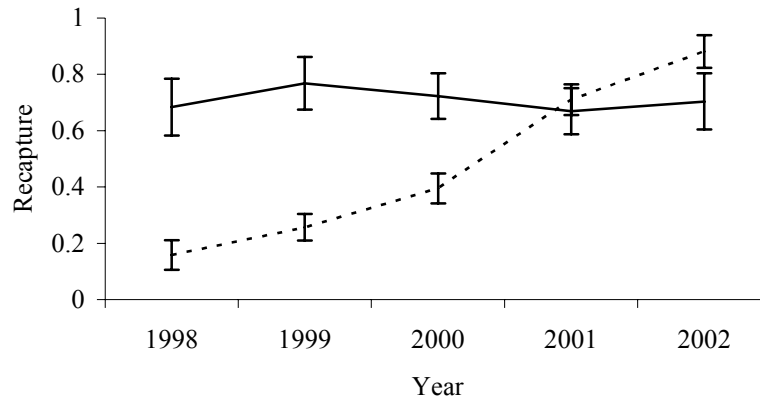


Figure 2.3 Recapture probabilities (and Standard Error, SE) of Balearic shearwaters during the study: solid line shows the recaptures at Conills colony and dashed lines recaptures at Sa Cella colony.

Using the demographic parameters estimated or set in our study (see Table 2.4), the results of the deterministic population model showed a declining trend ($\lambda = 0.952$), and population growth rate was significantly less than 1 ($z = 2.17$, $p < 0.01$). The growth rate was most sensitive to changes in adult survival, whereas the other parameters had a similar low impact on λ (Figure 2.4). Monte Carlo simulations showed that population trajectories decreased rapidly with time (Figure 2.5), at the same rate but inversely than extinction probabilities (Table 2.5). Demographic stochasticity played a more important role than environmental stochasticity in population growth rate (Table 2.5). With both stochasticities considered together, mean extinction time was computed at 40.4 years (SE = 0.2), and mean growth rate showed a 7.41% decrease per year (Table 2.5). When adult survival was set to a higher value (0.90) more normal for *Puffinus* species (e.g. Perrins et al. 1973, Warham 1990), the mean trajectory increased slowly ($\lambda = 1.0067$, SE = 0.001) (Figure 2.5) and none of the population trajectories went extinct ($z = 6.70$, $p < 0.001$).

Table 2.4 Estimate and 95% confidence intervals (CI) for demographic parameters used in the population models.

	Estimate	95% CI
Adult survival	0.78	[0.74-0.82]
Juvenile survival	0.70	[0.63-0.78]
Immature survival	0.74	[0.66-0.81]
Breeding success	0.59	[0.51-0.67]
Probability of skipping	0.26	[0.14-0.38]

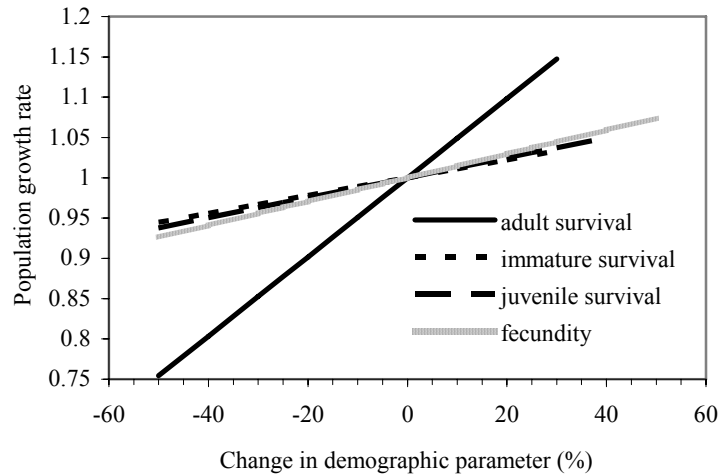


Figure 2.4 Sensitivity of the population growth rate (λ) to changes in the various demographic parameters of the deterministic model for Balearic shearwaters. Lines are truncated at the right when parameters attained their maximum values (for instance, adult survival because a 28% increase resulted in 100% survival).

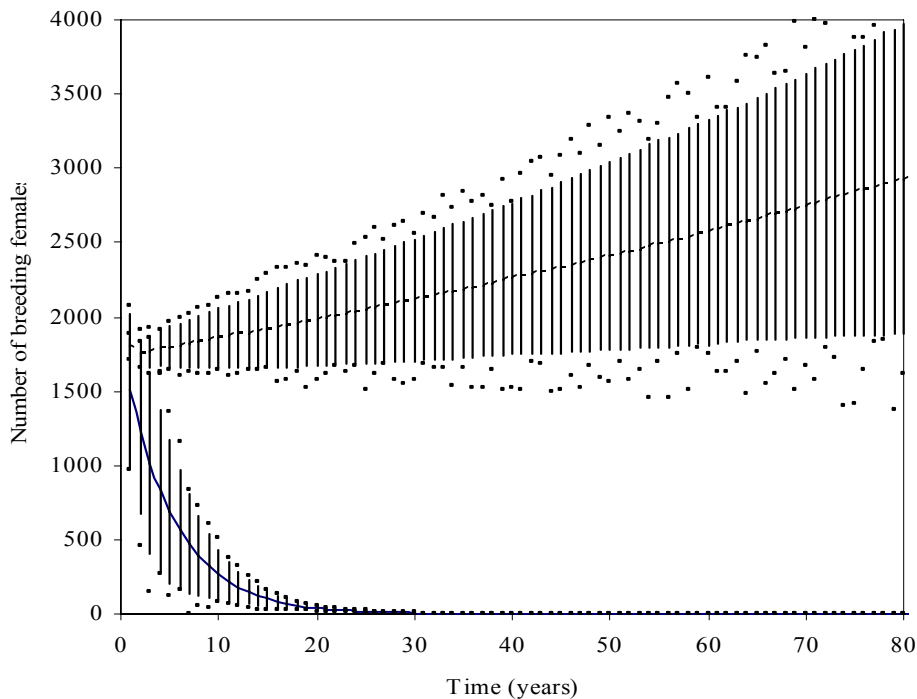


Figure 2.5 Simulated Monte Carlo population trajectories of Balearic shearwaters for the whole population. Solid line shows the mean trajectory with estimated adult survival at the study colonies, and dashed line with a theoretical adult survival for the species of 0.90. Bars represent two times the standard deviation (2σ) of the mean, and dots the maximum and minimum population value for each time step.

Table 2.5 Extinction probabilities for Balearic shearwater population simulations under the three scenarios considered: demographic and environmental stochasticity together, only demographic stochasticity, and only environmental stochasticity. Population growth rate λ (and standard error in parentheses) is shown for the three scenarios.

Year of simulation	Extinction probabilities		
	Environmental and demographic	Only demographic stochasticity	Only environmental stochasticity
10	0.0000	0.0000	0.0000
20	0.0000	0.0000	0.0000
30	0.0150	0.0000	0.0000
40	0.5610	0.0000	0.0000
50	0.9520	0.0200	0.0000
60	0.9980	0.3750	0.0000
70	1.0000	0.8150	0.0000
80	1.0000	0.9720	0.0250
90	1.0000	0.9950	0.1020
100	1.0000	0.9990	0.1524
λ	0.926 (0.001)	0.949 (0.001)	0.974 (0.0035)

2.5 Discussion

Results shown in our study confirm that a general decline is probably occurring for the whole population of Balearic shearwaters (Arcos & Oro 2004). In Cabrera Island, 60% of the colonies have disappeared in the last few decades (Aguilar 2000), while a recent survey in Formentera Island (where ca. 50% of the world population breeds) did not record any breeder in all the 30 suitable caves (where remains of ancient breeding were found) visited in a single cliff (SEO, unpubl. data). The lack of a reliable estimate of population size for the total world breeding population of Balearic shearwaters is one of the main problems when any population modelling is carried out. Even though numbers of Balearic shearwaters were underestimated (which could be the case especially for the non-breeding population) time to extinction would be simply delayed (Wissel & Zschke 1994). The degree of uncertainty is important for the reliability of population modelling (Hunter et al. 2000, Caswell 2001, Brook 2000, Williams et al. 2002). However, population growth rates and probabilities of extinction are independent of initial population sizes above a numeric threshold for which demographic stochasticity does not play a crucial role (Wissel & Zschke 1994). In our study reliability of extinction probabilities and growth rates are probably high because the most sensitive parameter to population growth rate in the species, adult survival, was estimated with a narrow confidence interval, which suggests that predictions of the models were sufficiently accurate (Brook 2000, Hunter et al. 2000). When breeding numbers are

difficult to estimate as is the case for many Procellariiformes (Gilbert et al. 1998, Heaney et al. 2002), estimation of demographic parameters and their use in population modelling are crucial to assess trends and to elaborate a conservation diagnose of endangered species (Bibby 1994, Beissinger & Westphal 1998, Russell 1999, Caswell 2001). Moreover, the simulations included the different values of demographic parameter estimates, so the best case scenarios were also considered. However, more years of monitoring would provide more robust data on the effects of age on breeding (e.g. Hunter et al. 2000) or on other parameters with a degree of uncertainty. Incorporating more data would also yield more robust survival estimates or would identified some environmental (climate, food) or individual (sex, age) features influencing survival (e.g. Tavecchia et al. 2001, Weimerskirch 2002). Large data sets have allowed researchers to refute the general thought that adult survival in seabirds was constant, and several studies have shown that survival can vary with time and with environmental factors especially during catastrophic events (e.g. climatic and oceanographic anomalies) or with humans activities (e.g. industrial fisheries, oil spills) (see revision in Weimerskirch 2002).

Results also suggest that the main conservation concern for Balearic shearwaters is adult survival, which is unusually low for a Procellariiform (e.g. Warham 1990, Hamer et al. 2002, Weimerskirch 2002). This demographic parameter should thus be managed to avert extinction, as is found in most long lived endangered species (e.g. Basse et al. 1997, Owens & Bennett 2000). Other demographic parameters such as productivity or skipping breeding seemed normal for a shearwater, although age of first reproduction (3 years) was younger than in most Procellariiformes. The most likely sources of adult mortality are also the main conservation concern for most seabirds throughout the world: at breeding sites, alien carnivores, and at feeding grounds, by-catch on fishing gears (mainly long lines) (Weimerskirch et al. 1997, Brothers et al. 1999, Tasker et al. 2000, Gilman 2001). Predators are probably the cause of historical extinction of Balearic shearwaters on the main islands (e.g. Alcover 2000) and are nowadays affecting large colonies such as those at Formentera Island (Aguilar 1997). The main predators are feral cats (Aguilar 2000, Mayol et al. 2000), a problem that affects many faunas at oceanic islands, where most recent extinctions have occurred (e.g. Bloomer & Bester 1992, Bibby 1994, Huyser et al. 2000, Risbey et al. 2000, Keitt et al. 2002). It is important to note that adult survival in our study was estimated in two colonies free of carnivores, and thus mortality was probably resulting from at-sea factors, and more

likely fishing gears. Mortality of Balearic shearwaters in long lines at the western Mediterranean is already suspected (Belda & Sánchez 2001, Cooper et al. 2003), but its impact is still unknown. Nothing is either known about this threat during migration and at wintering quarters, in regions where fisheries are important (e.g. south-western North Atlantic, western Africa), nor about other factors such as oil spills or drops in food availability that affect other shearwaters (Brichetti et al. 2000, Baduini et al. 2001, Lovvorn et al. 2001). Demographic stochasticity seems to play an important role in population dynamics of Balearic shearwaters (see Table 2.5), and more important than environmental stochasticity (Caswell 2000, 2001). Factors increasing adult mortality of Balearic shearwaters are probably also affecting other vulnerable seabird species breeding in the Mediterranean and wintering in the Atlantic Ocean, such as Levantine *P. yelkouan* and Cory's shearwaters *Calonectris diomedea*, European storm petrels *Hydrobates pelagicus* or Audouin's gull *Larus audouinii*.

Balearic shearwater is not yet among the list of most threatened European birds (Gallo-Orsi 2001). Political and administrative acceptance of taxonomic and conservation status by international conservation agencies is first needed to implement an effective action plan for the species. Population viability analysis performed in our study showed that probabilities of extinction are extremely high. Extinction time estimated here classified the species as critically endangered by IUCN (International Union for Conservation of the Nature) criteria, the highest category for a species non extinct in the wild. Urgent management measures (such as control of predators and reduction of mortality at fishing gears) and more years of monitoring to identify other threats are needed to ensure the conservation of Balearic shearwaters and in turn of other seabird species endemic of the Mediterranean region.

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References

- Aguilar JS (1992) Resum de l'atlas d'ocells marins de les Balears, 1991. Anuari Ornitologic Balear 7:17-28
- Aguilar JS (1993) Seguiment de la població reproductora de *Puffinus yelkouan mauretanicus* al Parc Nacional de Cabrera, 1993. Anuari Ornitologic Balear 8:51-53
- Aguilar JS (1997) Biología y conservación de la pardela balear *Puffinus mauretanicus*. Documents Tècnics de Conservació. Conselleria de Medi Ambient, Ordenació del Territori i Litoral (Govern Balear), Palma de Mallorca
- Aguilar JS (2000) La población de Pardela Balear (*Puffinus mauretanicus*) en el Parque Nacional del archipiélago de Cabrera. In: Pons GX (Ed) Las aves del Parque Nacional marítimo-terrestre del archipiélago de Cabrera (Islas Baleares, España). GOB-Ministerio de Medio Ambiente, Madrid, pp. 33-44
- Alcover JA (2000) Vertebrate evolution and extinction on western and central Mediterranean Islands. *Tropics* 10:103-123
- Amengual JF, Aguilar JS (1998) The impact of the black rat *Rattus rattus* on the reproduction of Cory's shearwaters *Calonectris diomedea* in the Cabrera National Park, Balearic Islands, Spain. In: Association "Les amis des oiseaux", Medmaravis (Eds) Ecologie des oiseaux marins et gestion intégrée du littoral en Méditerranée. Arcs Editions, Radès, pp. 94-121
- Anderson DR, Burnham KP, White CG (1994) AIC model selection in overdispersed capture-recapture data. *Ecology* 75:1780-1793
- Anderson DR, Burnham KP (1999) Understanding information criteria for selection among capture-recapture or ring-recovery models. *Bird Study* 46:14-21
- Arcos JM, Oro D (2002) Significance of fisheries discards for a threatened Mediterranean seabird, the Balearic shearwater *Puffinus mauretanicus*. *Marine Ecology Progress Series* 239:209-220
- Arcos JM, Massutí E, Abelló P, Oro D (2000) Fish associated with floating drifting objects as a feeding resource for Balearic Shearwaters *Puffinus mauretanicus* during the breeding season. *Ornis Fennica* 77:177-182

- Arcos JM, Oro D (2004) Pardela Balear *Puffinus mauretanicus*. In: Madroño A, Martí R (Eds) Libro Rojo de las Aves de España. Dirección General para la Biodiversidad- SEO/BirdLife, Madrid, pp. 46-50
- Austin JJ (1996) Molecular phylogenetics of *Puffinus* Shearwaters: Preliminary evidence from mitochondrial Cytochrome b gene sequences. *Molecular Phylogenetics and Evolution* 6:77-88
- Baduini CL, Lovvorn JR, Hunt GLJr (2001) Determining the body condition of short-tailed shearwaters: implications for migratory flight ranges and starvation events. *Marine Ecology Progress Series* 222:265-277
- Basse B, Wake GC, McLennan JA (1997) Predation thresholds for survival of endangered species. *IMA Journal of Mathematics Applied in Medicine and Biology* 14:241-250
- Beissinger SR, Westphal MI (1998) On the use of demographic models of population viability in endangered species management. *Journal of Wildlife Management* 62:821-841
- Belda EJ, Sánchez A (2001) Seabird mortality on longline fisheries in the western Mediterranean: factors affecting bycatch and proposed mitigating measures. *Biological Conservation* 98:357-363
- Bibby CJ (1994) Recent past and future extinctions in birds. *Philosophical Transactions of the Royal Society of London B Biological Sciences* 344:35-40
- Bloomer JP, Bester MN (1992) Control of feral cats on sub-Antarctic Marion Island, Indian Ocean. *Biological Conservation* 60:211-219
- Brichetti P, Foscolo U, Boano G (2000) Does El Niño affect survival rate of Mediterranean populations of Cory's shearwater? *Waterbirds* 23:147-154
- Brook BW (2000) Pessimistic and optimistic bias in population viability analysis. *Conservation Biology* 14:564-566
- Brothers N, Cooper J, Lokkeborg S (1999) The incidental catch of seabirds by longline fisheries: worldwide review and technical guidelines for mitigation. Unpublished Report. FAO Fisheries Circular, Rome
- Burnham KP, Anderson DR (1998) *Model Selection and Inference, a Practical Information-Theoretic Approach*. Springer-Verlag, New York
- Burnham KP, White GC (2002) Evaluation of some random effects methodology applicable to bird ringing data. *Journal of Applied Statistics* 29:245-264

- Burnham KP, Anderson DR, White GC (1995) Selection among open population capture-recapture models when capture probabilities are heterogeneous. *Journal of Applied Statistics* 22:611-624
- Burnham KP, Anderson DR, White GC, Brownie C, Pollock KH (1987) Design and analysis methods for fish survival experiments based on release-recapture. American Fisheries Society, Bethesda
- Caswell H (2000) Prospective and retrospective perturbation analysis: their roles in conservation biology. *Ecology* 81:619-627
- Caswell H (2001) Matrix population models. Sinauer Associates, Sunderland, USA
- Choquet R, Reboulet A-M, Pradel R, Lebreton J-D (2000) U-care (Utilities - Capture-REcapture) user's guide. CEFE-CNRS, Montpellier (<ftp://ftp.cefe.cnrs-mop.fr/biom/Soft-CR/u-care/>)
- Cooper J, Baccetti N, Belda EJ, Borg JJ, Oro D, Papaconstantinou C, Sánchez A (2003) Seabird mortality from longline fishing in the Mediterranean Sea and Macaronesian waters: a review and a way forward. *Scientia Marina* 67S2:57-64
- Croxall JP, Rothery P (1991) Population regulation of seabirds: implications of their demography for conservation. In: Perrins CM, Lebreton J-D, Hirons GJM (Eds) *Bird Population Studies, Relevance to Conservation and Management*. Oxford University Press, Oxford, pp. 272-296
- Gallo-Orsi U (2001) Saving Europe's most threatened birds: progress in implementing European Species Action Plans. Unpublished Report. BirdLife International, Wageningen
- Genovart M, McMinn M, Bowler D (2003) A discriminant function for predicting sex in the Balearic Shearwater. *Waterbirds* 26:72-76
- Gilbert G, Gibbons DW, Evans J (1998) *Bird Monitoring Methods*. RSPB, Sandy
- Gilman E (2001) Integrated management to address the incidental mortality of seabirds in longline fisheries. *Aquatic Conservation* 11:391-414
- Hamer KC, Schreiber EA, Burger J (2002) Breeding biology, life histories, and life history-environment interactions in seabirds. In: Schreiber EA, Burger J (Eds) *Biology of Marine Birds*. CRC Press, Boca Raton, pp. 217-261
- Harris MP, Buckland ST, Russell SM, Wanless S (1994) Post fledging survival to breeding age of Shags *Phalacrocorax aristotelis* in relation to year, date of fledging and brood size. *Journal of Avian Biology* 25:268-274

- Heaney V, Ratcliffe N, Brown A, Robinson PJ, Lock L (2002) The status and distribution of European Storm-petrels *Hydrobates pelagicus* and Manx Shearwaters *Puffinus puffinus* on the isles of Scilly. *Atlantic Seabirds* 4:1-16
- Heidrich P, Amengual JF, Wink M (1998) Phylogenetic relationships in Mediterranean and North Atlantic shearwaters (Aves: Procellariidae) based on nucleotide sequences of mtDNA. *Biochemical Systematics and Ecology* 26:145-170
- Heidrich P, Amengual JF, Ristow D, Wink M (2000) Phylogenetic relationships among Procellariiformes based on nucleotide sequences, with special consideration of the Mediterranean and North Atlantic Shearwaters. In: Yésou P, Sultana J (Eds) *Monitoring and Conservation of Birds, Mammals and Sea Turtles of the Mediterranean and Black Seas*. Environment Protection Department, Malta, pp. 159-175
- Heppell SS, Caswell H, Crowder LB (2000) Life histories and elasticity patterns: perturbation analysis for species with minimal demographic data. *Ecology* 81: 654-665
- Hines JE (1996) TMSURVIV User's Manual. Pages National Biological Survey, Patuxent Wildlife Research Center, Laurel
- Hunter CM, Moller H, Fletcher D (2000) Parameter uncertainty and sensitivity analysis of a population model: setting research priorities for shearwaters. *Ecological Modelling* 134:299-323
- Huysen OAW, Ryan PG, Cooper J (2000) Changes in population size, habitat use and breeding biology of Lesser Shearwaters (*Chionis minor*) at Marion Island: impacts of cats, mice and climate change? *Biological Conservation* 92:299-310
- Keitt BS, Wilcox C, Tershy BR, Croll DA, Donlan CJ (2002) The effect of feral cats on the population viability of black-vented shearwaters (*Puffinus opisthomelas*) on Natividad Island, Mexico. *Animal Conservation* 5:217-223
- Lebreton J-D (2001) The use of bird rings in the study of survival. *Ardea* 89:85-100
- Lebreton J-D, Clobert J (1991) Bird population dynamics, management, and conservation: the role of mathematical modelling. In: Perrins CM, Lebreton J-D, Hiron GJM (Eds) *Bird Population Studies, Relevance to Conservation and Management*. Oxford University Press, Oxford, pp. 105-125
- Lebreton J-D, Hemery G, Clobert J, Coquillart H (1990) The estimation of age-specific breeding probabilities from recaptures or resightings in vertebrate populations. I. Transversal models. *Biometrics* 46:609-622

- Lebreton J-D, Burnham KP, Clobert J, Anderson DR (1992) Modeling survival and testing biological hypothesis using marked animals: a unified approach with case studies. *Ecological Monographs* 62:67-118
- Lovvorn RJ, Baduini CL, Hunt GLJr (2001) Modeling underwater visual and filter feeding by planktivorous shearwaters in unusual sea conditions. *Ecology* 82:2342-2356
- Mayol J (1986) Human impact on seabirds in the Balearic Islands. In: Medmaravis, Monbailu X (Eds) *Mediterranean Marine Avifauna: population studies and conservation*. Springer-Verlag, Berlin, pp. 379-396
- Mayol J (1997) Specific identity of the Balearic Shearwater *Puffinus mauretanicus* (Lowe 1951). *Anuari Ornitologic Balear* 12:3-11
- Mayol J, Aguilar JS, Yésou P (2000) The Balearic Shearwater *Puffinus mauretanicus*: status and threats. In: Yésou P, Sultana J (Eds) *Monitoring and Conservation of Birds, Mammals and Sea Turtles of the Mediterranean and Black Seas*. Environment Protection Department, Malta, pp. 24-37
- Nichols JD, Spendelow JA, Hines JE (1990) Capture-recapture estimation of prebreeding survival rate for birds exhibiting delayed maturation. *Journal of Field Ornithology* 61:347-354
- Oro D, Pradel R, Lebreton J-D (1999) The effects of nest predation and food availability on life history traits in Audouin's gull. *Oecologia* 118:438-445
- Owens IPF, Bennett PM (2000) Ecological basis of extinction risk in birds: habitat loss versus human persecution and introduced predators. *Proceedings of the National Academy of Science* 97:12144-12148
- Perrins CM, Harris MP, Britton CK (1973) Survival of Manx shearwaters *Puffinus puffinus*. *Ibis* 115:535-548
- Pollock KH, Nichols JD, Brownie C, Hines JE (1990) Statistical inference for capture-recapture experiments. *Wildlife Monographs* 107
- Pradel R, Hines JE, Lebreton J-D, Nichols JD (1997) Capture-recapture survival models taking account of transients. *Biometrics* 53:88-99
- Risbey DA, Calver MC, Short J, Bradley JS, Wright IW (2000) The impact of cats and foxes on the small vertebrate fauna of Heirisson Prong, Western Australia. II. A field experiment. *Wildlife Research* 27:223-235
- Russell RW (1999) Comparative demography and life history tactics of seabirds: implications for conservation and marine monitoring. In: Musick JA (Ed) *Life in*

- the slow lane: ecology and conservation of long-lived marine animals. American Fisheries Society, Bethesda, pp. 51-76
- Spendelov JA, Nichols JD, Hines JE, Lebreton J-D, Pradel R (2002) Modelling postfledging survival and age-specific breeding probabilities in species with delayed maturity: a case study of Roseate Terns at Falkner Island, Connecticut. *Journal of Applied Statistics* 29:385-405
- Tasker ML, Camphuysen CJ, Cooper J, Garthe S, Montevecchi WA, Blaber SJM (2000) The impacts of fishing on marine birds. *ICES Journal of Marine Science* 57:531-547
- Tavecchia G, Pradel R, Boy V, Johnson AR, Cézilly F (2001) Sex- and age-related variation in survival and the cost of first reproduction in Greater Flamingos. *Ecology* 82:165-174
- Warham J (1990) *The Petrels. Their Ecology and Breeding Systems*. Academic Press, London, UK
- Weimerskirch H (2002) Seabird demography and its relationship with the marine environment. In: Schreiber EA, Burger J (Eds) *Biology of Marine Birds*. CRC Press, Boca Raton, pp. 115-135
- Weimerskirch H, Brothers N, Jouventin P (1997) Population dynamics of wandering albatross *Diomedea exulans* and Amsterdam Albatross *D. amsterdamensis* in the Indian Ocean and their relationships with long-line fisheries: conservation implications. *Biological Conservation* 79:257-270
- White GC, Burnham KP (1999) Program MARK: survival estimation from populations of marked animals. *Bird Study* 46:120-139
- Williams BK, Nichols JD, Conroy MJ (2002) *Monitoring and Management of Animal populations*. Academic Press, San Diego
- Wissel Ch, Zashke SH (1994) Stochastic birth and death processes describing minimum viable populations. *Ecological Modelling* 76:193-201
- Yearsley JM, Fletcher D (2002) Equivalence relationships between stage-structured population models. *Mathematical Biosciences* 179:131-143
- Yearsley JM, Fletcher D, Hunter C (2003) Sensitivity analysis of equilibrium population size in a density-dependent model for Short-tailed Shearwaters. *Ecological Modelling* 163:1-11

Chapter 3



Small pelagic fish, trawling discards and breeding performance of the critically endangered Balearic shearwater: improving conservation diagnosis

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3.1 Abstract

Conservation diagnosis should identify which demographic parameters (or vital rates) are having a negative effect on the population growth rate of a threatened organism. Diagnosis can analyse how the environment is influencing the vital rates of this organism. In the present study, we tried to improve the diagnosis of a critically endangered seabird, the Balearic shearwater *Puffinus mauretanicus*, through analysis of the spatio-temporal variability of breeding performance, using long-term data sets covering most of its breeding range. We also analysed the potential influence of two

food resources, namely small pelagic fish and trawling discards, as measured by purse-seine and trawling landings, respectively. We found inter-annual variability in breeding performance, which could be related to the changes in the availability of both small pelagic fish and trawling discards. Results showed that these variations affected all study colonies in a similar way, thus supporting previous investigations that identified a major foraging area for the whole breeding population. A fitness landscape suggested that current values of breeding success are not responsible for the sharp decline of the species, although incoming fisheries policies could affect its breeding performance in the long term. Factors affecting adult survival, both at sea and in colonies, remain the most important threat, and conservation efforts should concentrate on these issues to prevent the extinction of this endemic species of the Mediterranean region.

3.2 Introduction

The estimation of demographic parameters is one of the most powerful and reliable tools in diagnosing the level of conservation concern of a species (Krebs 2001). Robustness and reliability of matrix population models in population viability analysis mainly depend on good, unbiased estimators of these demographic parameters (e.g. Saether & Engen 2002). Conservation diagnosis is directed to identify which demographic parameter is negatively affecting the population growth rate of a threatened organism (Krebs 2001, Norris 2004). Diagnoses are always phrased in terms of differences (or comparisons) in the vital rates in both space and time (Caswell 2001), for instance, by using phylogenetically close and well-known species with similar life histories. Diagnosis also analyses how the environment is influencing the vital rates of a species.

Over the past centuries, changes in the environment caused by human activities have had a negative effect on the functioning of ecosystems and the population dynamics of many organisms (see Botsford et al. 1997). This is especially true for species occupying the top level of trophic chains, where environmental changes are amplified.

In marine ecosystems, one of these human activities (i.e. fishery) is responsible for the decline of marine apical species, removing 90% of large predatory fishes, and is also blamed for an 80% reduction of the community biomass (Myers & Worm 2003, Lewison et al. 2004). At the same time, many marine predators with high behavioural

plasticity take advantage of fisheries by scavenging on discards and offal (Oro 1999, Tasker et al. 2000). Opportunistic seabirds are good representatives of these organisms, and their populations are showing important changes in numbers resulting from a balance of positive and negative effects due to the birds' interaction with fisheries, such as exploitation of discards and mortality in long lines, respectively (reviewed in Montevecchi 2002, see also Tasker et al. 2000, Lewison et al. 2004).

The critically endangered Balearic shearwater *Puffinus mauretanicus*, endemic to the Mediterranean region, is an example of a marine top predator specialised in foraging on shoals of small pelagic fishes, which currently interacts in a number of ways with fisheries (Oro & Ruiz 1997, Arcos & Oro 2004). A bioenergetics model estimated that >40% of the energetic demands of the world population of this species are met by trawling discards (Arcos & Oro 2002). Oro et al. (2004) estimated an unusually low adult survival rate, which would be unsustainable for such a long-lived organism, and suggested that entanglement in fishing gear could be the main cause of adult mortality. In the latter study, population modelling showed a declining trend for the world breeding population ($\hat{\lambda} = 0.926$, SE = 0.001), and mean extinction time was estimated at 40.4 yr (SE = 0.2). In contrast to adult survival, little is known about the potential relevance of other important demographic parameters for the conservation of the Balearic shearwater, such as breeding performance.

Introduced predators such as rats and carnivores, abundant on Mediterranean islands, can negatively affect the breeding performance of shearwaters and other organisms (Martin et al. 2000). On the other hand, incoming fishing policies, aimed at decreasing both fishing efforts (Pauly et al. 2003) and the amounts of discarded fish (Fluharty 2000), are expected to have the opposite effect, with an unknown balance resulting from: (1) a decrease of discard availability and, in turn, an influence on the breeding performance of Balearic shearwaters and (2) an increase in pelagic fish for seabirds following ecosystem restoration.

Taking into account the critical status of Balearic shearwaters, conservation diagnosis can be improved by analysing their breeding performance and the influence of the environment, especially the availability of both small pelagic fish and discards, on this parameter. Here, we were able to compile a long-term historical data set (up to 15 yr) from six colonies, covering most of the breeding range of the species. In the case of threatened species, the information available is usually limited to a very small number of colonies and years. However, we decided to use the most consistent time series (i.e.

1997 to 2004) to perform modelling (see Table 3.1). Our main aims were: (1) to analyse the spatio-temporal variability of breeding performance in Balearic shearwaters, (2) to investigate the influence of fisheries landings as a proxy for food resource availability and (3) to assess the contribution of the breeding success to the population viability of the species.

Table 3.1 Hatching and breeding success of Balearic shearwaters in six colonies from 1986 through 2004. Sample sizes (number of nests monitored) are shown in brackets.

Breeding parameter	Year	Cabrera	Conillera	Conills	Malgrats	Maó	Sa Cella
Hatching success	1987			0.69 (16)			
	1993	0.44 (9)					
	1994	0.60 (5)					
	1995	0.57 (7)					
	1997	0.56 (16)		0.93 (16)			0.7 (54)
	1998	0.75 (16)		0.81 (16)			0.50 (42)
	1999	1.00 (9)	0.71 (7)	0.55 (20)	0.67 (6)	0.78 (9)	0.71 (24)
	2000	0.80 (10)		0.75 (16)	0.93 (15)	0.89 (19)	0.92 (25)
	2001			0.56 (23)	0.62 (21)		0.68 (111)
	2002			0.58 (26)		0.90 (10)	0.61 (94)
	2003			0.74 (23)		0.60 (15)	0.74 (70)
	2004			0.52 (23)		0.5 (10)	0.56 (89)
	Breeding success	1986			0.53 (17)	0.71 (17)	
1987				0.56 (16)	0.33 (18)		
1988				0.67 (18)			
1993		0.44 (9)					
1994		0.6 (5)					
1995		0.57 (7)					
1997		0.56 (16)		0.81 (16)	1.00 (7)		0.65 (54)
1998		0.69 (16)		0.69 (16)			0.45 (42)
1999		0.89 (9)	0.71 (7)	0.45 (20)	0.67 (6)	0.67 (9)	0.68 (24)
2000		0.8 (10)		0.62 (16)	0.93 (15)	0.84 (19)	0.88 (25)
2001				0.56 (23)	0.62 (21)	0.60 (10)	0.67 (110)
2002				0.58 (26)		0.80 (10)	0.53 (94)
2003				0.74 (23)		0.60 (15)	0.73 (70)

3.3 Material and Methods

Species details and study area

The Balearic shearwater is a medium-sized Procellariiforme, with an adult body mass of ca. 500 g. The species shares most life-history traits with other members of the family: lays a single egg, has long incubation and chick-rearing stages and large and distant foraging areas. Balearic shearwaters have a small breeding population (ca. 2000 pairs) and are restricted to the Balearic archipelago (Ruiz & Martí 2004), both situations increasing their conservation concern (Krebs 2001).

The study was conducted from 1986 through 2004 at the 6 main known and accessible colonies. Surveyed colonies (Figure 3.1) were of different sizes, ranging from 40 to 300 breeding pairs, and were located in deep caves and between alluvia in coastal cliffs or islets. The introduction of alien species, human harvesting, and coastal development have all contributed to a reduction in breeding habitat (Arcos & Oro 2004, Ruiz & Martí 2004).

Breeding performance

We visited the colonies at the beginning of the breeding season (mid-March) and marked all accessible nests in order to monitor egg laying. Visits were repeated in mid-May to record the number of hatchlings and in mid-June to count fledglings at each marked nest. Hatching success was defined as the number of chicks hatched (mid-April to early May) divided by the total number of eggs laid at the beginning of the breeding period (late February to early March). Breeding success was calculated as the number of chicks fledged (mid-June to early July) divided by the number of eggs laid.

Fisheries data

The Ebro delta area (one of the main foraging grounds where shearwaters forage during the breeding season; Arcos & Oro 2002, Ruiz & Martí 2004) is a favourable spawning area for many fish species (Agostini & Bakun 2002), especially small pelagic fish, and supports one of the largest fishing fleets in the western Mediterranean (e.g. Abad et al. 1998). Catches of anchovy *Engraulis encrasicolus* in this area are the highest

for the Mediterranean (Agostini & Bakun 2002). The high primary productivity of the area, in the Mediterranean context, is due to the freshwater input from the Ebro River, the strong winds and a relatively wide continental shelf, coupled with the influence of the Liguro–Porvençal–Catalan front at the continental slope (Salat et al. 2002).

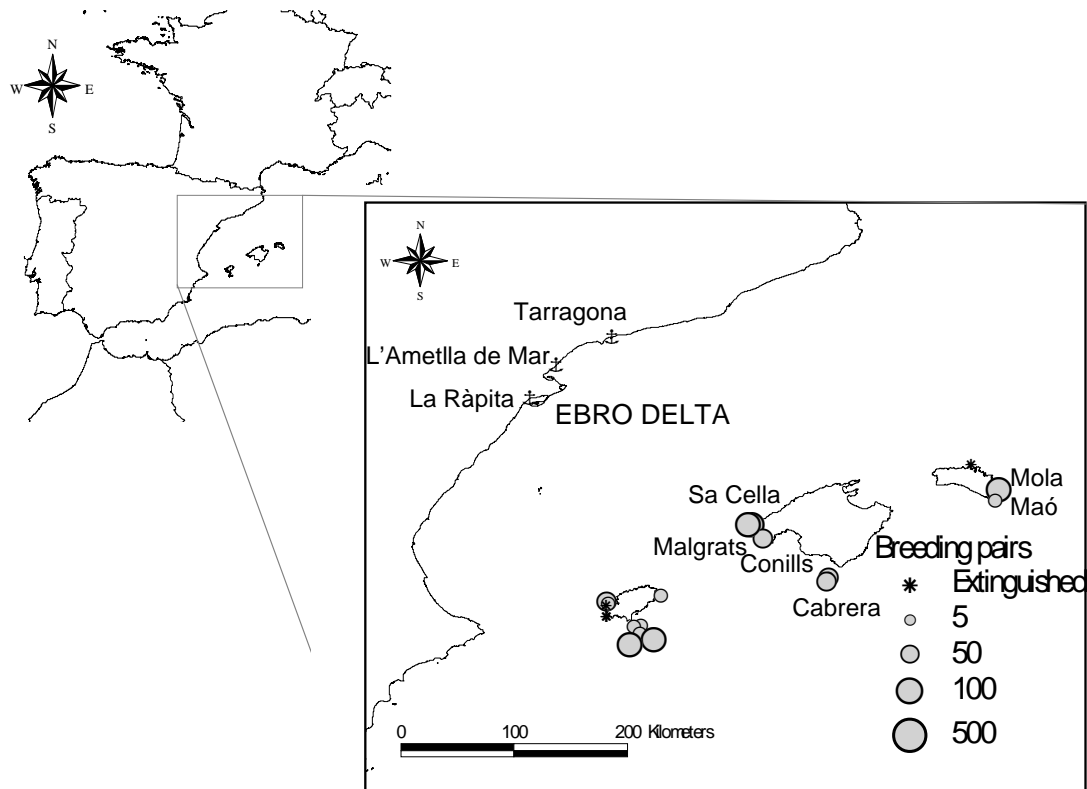


Figure 3.1 Map of the study area, showing the Balearic shearwater colonies and their size. Monitored colonies and main fishing harbours mentioned throughout the text are also shown.

We used statistics of fisheries landings from this area (www.gencat.net/darp/c/pescamar/captures/cestca03.htm) as an index of food resource availability (Montevecchi 2002). In the Ebro delta, there is some evidence that trawling and purse-seine landings are reliable indices of discard availability and small pelagic fish abundance, respectively (Oro 1999, Oro et al. 1999, Lloret et al. 2004). Data on small pelagic fish landings have been used successfully to perform stock biomass assessments and recruitment analyses in the Mediterranean (Santojanni et al. 2003, Lloret et al. 2004). Also, anchovy landings accounted for 38 to 55% of the biomass estimated by acoustic surveys in the NW Mediterranean in the early 1990s (Abad et al. 1998). Thus, we assumed that small pelagic fish landings could be used as a proxy for their availability for Balearic shearwaters. Concerning trawling discards in the Ebro Delta, Oro & Ruiz (1997) found a positive association between the amounts of fish

discarded and landed varying between 15 and 45%. Moreover, Arcos (2001) computed a discard ratio (i.e. relation between discard and landing) representing, on average, slightly more than one-third of total catches (36%, given the mean discard ratio of 56%) in the same area. Also, in the NW Mediterranean, discards of the trawl fleet represent ca. 40% of the total catch on the continental shelf (Carbonell et al. 1998). All studies were performed in different years and estimated a similar positive relationship between discarded and landed amounts of fish or total catch. Therefore, we assumed that this positive association was consistent throughout the analysed time period. However, trawlers' landing data have to be used with caution, since, in some cases, the relation between discards and catches is not proportional (Rochet & Trenckel 2005).

These landings were taken from the 3 main fishing harbours which represented ca. 80% of the total landings in the province of Tarragona (Catalonia, see Figure 3.1). L'Ametlla de Mar and Tarragona were taken as representative of the purse-seine catches, and Sant Carles de la Ràpita, as representative of trawling catches, since >90% of the landings in these harbours corresponded to these types of fishing gear from 2000 through 2004. We analysed the potential association of landings with hatching and breeding success of the shearwaters by grouping the landings by month according to the breeding cycle of the species, with January and February being the pre-laying period, March and April the incubation period and March to June the overall breeding season.

Statistical analysis

We used generalized linear models (GLMs, GENMOD procedure in SAS 2000) (1) to assess the influence of year, colony and their interaction on hatching and breeding success of shearwaters and (2) when testing the relationship between both breeding performance and food resource availability. We considered data from 5 colonies (Cabrera, Conills, Malgrats, Mola Maó and Sa Cella; see Figure 3.1) collected during the 1997 to 2004 period and excluded previous years because data were too sparse (see Table 3.1). For these analyses, hatching and breeding success were treated as binomial dependent variables and fitted using a logit-link function. We evaluated the goodness-of-fit of each model using the Pearson's chi-square statistic (Crawley 1993).

For spatio-temporal analysis of breeding performance, models were compared using Akaike's information criterion (AIC; see Williams et al. 2001). The model with the

lowest AIC is considered the best compromise between model deviance and model complexity, i.e. the number of parameters in the model. When models are within 4 points of AIC, they are considered statistically equivalent (Williams et al. 2001). In this case, we chose the model with fewest parameters, this being the most parsimonious. We tested the effects of year (t), colony (c), additive models ($c + t$) and their statistical interaction ($t * c$).

When analysing the influence of food resources, we log-transformed landings data. We hypothesized that: (1) both breeding parameters depended on the food availability during the pre-laying period, (2) hatching success depended on nest features (i.e. incubation period) and (3) breeding success was influenced by food availability during the overall breeding season. Significance level was set at 0.05, and contrast analyses were adjusted by Bonferroni correction.

To assess the contribution of breeding success to the population viability of the species, a fitness landscape was calculated using a deterministic population matrix model (Caswell 2001). The fitness landscape is a tool to show how variations of a pair of demographic parameters yield curves with a constant population growth rate λ (or population fitness). All parameters of the matrix, except fecundity (i.e. breeding success), were taken from Oro et al. (2004). For our purposes, the landscape was generated using breeding success (the parameter of interest here) and adult survival (the most sensitive parameter for the species). We used mean values (and 95% confidence intervals) of breeding success estimated at 5 colonies (Cabrera, Conills, Malgrats, Mola Maó and Sa Cella) from which we had at least 5 yr of monitoring data from the historical database (1986 to 2004). Breeding success is a demographic parameter of conservation concern when these distributions are below the curve with $\lambda = 1$, i.e. the curve of population stability.

Table 3.2 Modelling spatio-temporal variability in breeding parameters of the Balearic shearwater in 5 colonies from 1997 through 2004 (AIC: Akaike's information criterion; np: number of estimable parameters in the model). For model notation see 'Materials and methods'. Selected models are in bold.

Breeding parameter	Effect	Deviance	np	AIC
Hatching success	<i>t</i>	1016.108	8	1032.108
	<i>c</i>	1042.417	5	1052.417
	<i>t + c</i>	1015.528	12	1039.528
	<i>t + c + t × c</i>	983.806	28	1039.806
Breeding success	<i>t</i>	1070.082	8	1086.082
	<i>c</i>	1093.236	5	1103.236
	<i>t + c</i>	1067.641	12	1091.642
	<i>t + c + t × c</i>	1041.071	30	1101.071

3.4 Results

Breeding performance

We gathered information on 828 and 840 monitored nests to compute hatching and breeding success, respectively. Sa Cella was a unique colony, free of predators, whereas eradication campaigns have been carried out for several years at the other colonies, mostly because of rats. Hatching success values ranged from 0.50 to 1.00, while breeding success varied between 0.45 and 0.93 (Table 3.1). Analysis of the spatio-temporal variability of breeding performance only showed inter-annual variation in the 2 breeding parameters, whereas we did not find any colony effect (Table 3.2). The goodness-of-fit test indicated that models fitted the data correctly (for hatching success $\chi^2_{828} = 828.00$, $p = 0.58$; for breeding success $\chi^2_{832} = 840.00$, $p = 0.58$). Both breeding parameters showed a similar pattern and magnitude during the study period (Figure 3.2), suggesting that breeding failure occurs mostly during the incubation and hatching periods and not during the chick-rearing stage.

Influence of food resource availability on breeding performance

Availability of both small pelagic fish and trawling discards during pre-laying positively influenced overall breeding performance (see Table 3.3). More precisely, breeding success was significantly influenced by the availability of both resources, whereas hatching success was influenced only by discard availability. Both food resources showed a similar trend during pre-laying (see Figure 3.2a), and it was not

possible to disentangle their influences on hatching success. During incubation, the abundance of small pelagic fish was significantly associated with hatching success (see Table 3.3). In terms of the whole reproductive season, both small pelagic fishes and trawling discards, but particularly the former, influenced breeding success (see Table 3.3, Figure 3.2b). All goodness-of-fit tests indicated that models fitted the data adequately (i.e. all tests were non-significant; results not shown for clarity).

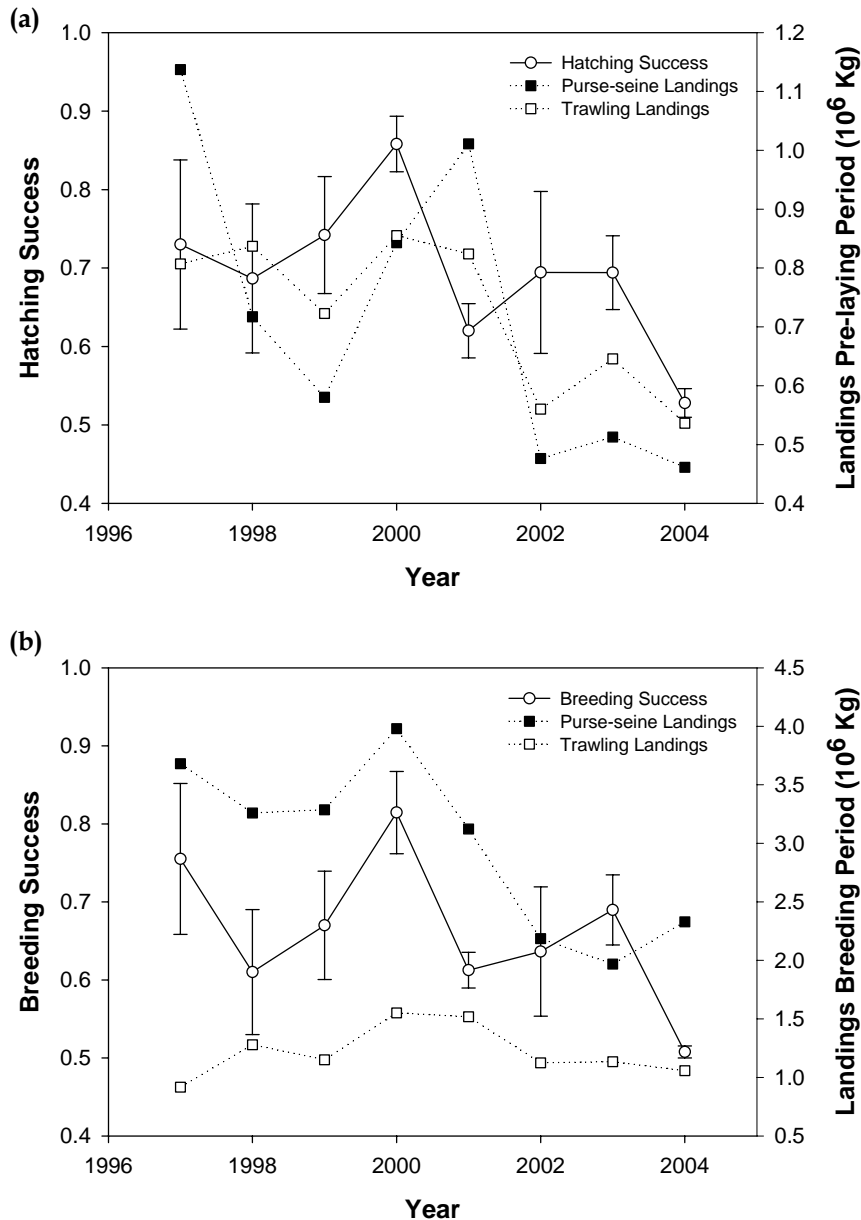


Figure 3.2 Inter-annual average of: (a) hatching and (b) breeding success (solid lines) with respect to landings of both fisheries (dashed lines) during pre-laying and breeding period, respectively. Note scale difference for both graphics on the right axis (landings)

Table 3.3 Influence of food resources on breeding performance of Balearic shearwaters. Purse-seine and trawling landings are used as proxies for small pelagic fish and discard availability, respectively. Landings were log-transformed. Significant relationships after applying Bonferroni correction are shown in bold. For breeding cycle explanation see 'Materials and methods'.

Breeding parameter	Fisheries	Breeding cycle			
		Pre-laying		Incubation	
		χ_1^2	p	χ_1^2	P
Hatching success	Purse-seine	5.26	0.022	6.74	0.009
	Trawling	10.09	0.001	3.6	0.058
		Pre-laying		Breeding	
		χ_1^2	p	χ_1^2	p
Breeding success	Purse-seine	7.26	0.007	6.04	0.014
	Trawling	10.71	0.001	3.94	0.047

Fitness landscape

Once the mean values of breeding success and their confidence intervals on the fitness landscape had been traced (and assuming that 0.9 is the typical adult survival for the species, Figure 3.3a), all distributions (each one corresponding to each monitored colony) showed values above the stability line (i.e. the line with $\lambda = 1$). With such demographic parameters, and considering 0.9 as a very conservative value for adult survival (see Oro et al. 2004), results suggest that with such breeding success, the population would never become extinct. When plotting the estimated actual value of adult survival (0.78), the fitness landscape showed that any conservation measure directed to enhance the breeding success should increase this parameter by ca. 20% to attain a mean success of 0.75 chicks per pair (Figure 3.3b), a value that was reached only in years of high food availability (see Table 3.1, Figure 3.2b).

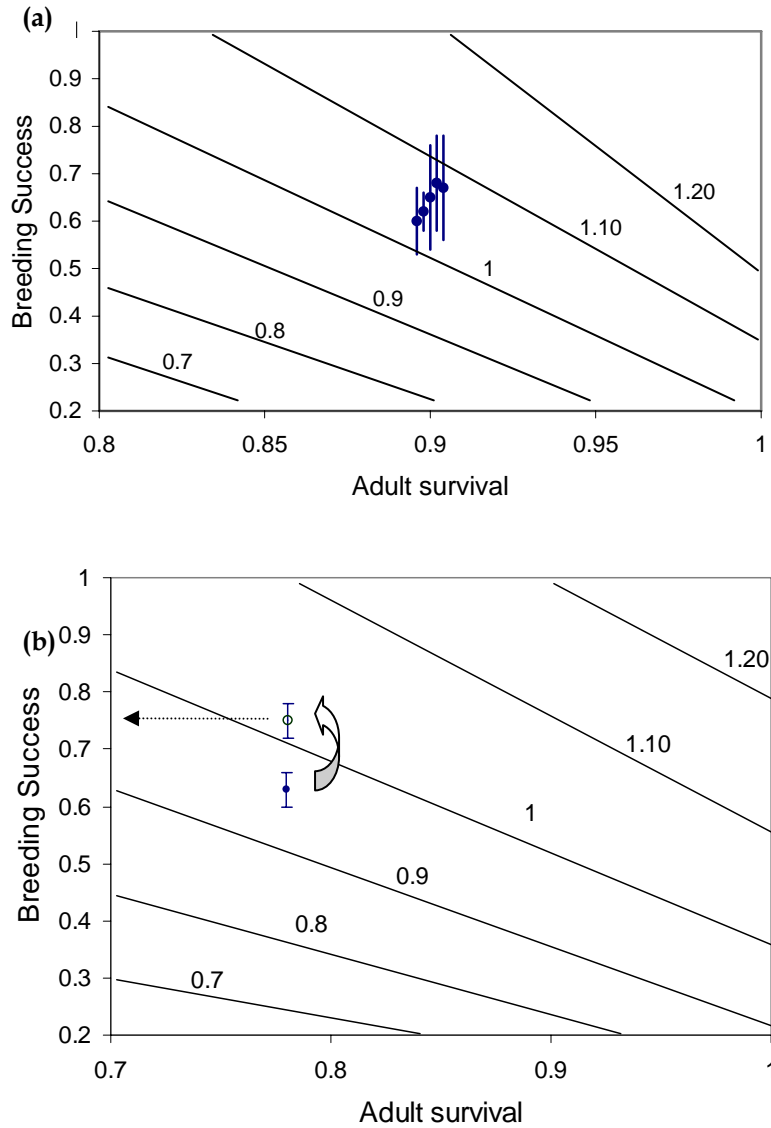


Figure 3.3 Fitness landscape of adult survival against breeding success, yielding curves with different values of population growth rate λ ($\lambda = 1$ represents the curve of population stability). (a) Mean values of breeding success (together with 95% confidence intervals) are shown from the 5 study colonies with sufficient data, assuming an adult survival of 0.9, see Oro et al. 2004). These values have been separated slightly in the figure for the sake of clarity; (b) actual value of estimated survival for the species (0.78 from Oro et al. 2004, shown by the solid dot) is plotted against the mean breeding success of the 5 study colonies (with its 95% confidence intervals); the arrow points to a hypothetical value of breeding success (shown by the open dot) necessary to exceed the curve of population stability. Error bars are 95% confidence intervals.

3.5 Discussion

Spatio-temporal variability in breeding performance and food resources

Our results show that the breeding performance of Balearic shearwaters *Puffinus mauretanicus* changed from year to year and that these variations affected all study colonies similarly. Breeding performance of Balearic shearwaters was influenced by the availability of both small pelagic fish (their main prey in natural conditions, i.e. in the absence of trawling fisheries) and trawling discards. The incubation period appears to be the most critical stage with regard to breeding failure; this could be influenced by the pre-breeding female condition through temporal variability in food availability (Barbraud & Chastel 1999). Temporal variability in the breeding performance of shearwaters was probably a reflection of annual changes in the availability of small pelagic fish, which exhibit stochastic temporal patterns. The life history of small pelagic fish (e.g. high mobility, plankton-based food chains and short life span) makes these species particularly sensitive to environmental stochasticity, with quick and dramatic responses to climatic and oceanographic variations (Guisande et al. 2001, Lloret et al. 2004). Our findings suggest that this variability is transported up the food chain, affecting top marine predators such as seabirds (see also Sydeman et al. 2001). Consistently, several studies have demonstrated the influence of climatic oscillations and other oceanographic features (e.g. sea surface temperature anomalies) in the inter-annual variability of the breeding performance of seabirds (Sydeman et al. 2001, Thompson & Ollason 2001).

In the case of trawling discards, our results support those of Arcos & Oro (2002) who found that >40% of the energetic requirements of Balearic shearwaters during the chick-rearing period are met from the exploitation of trawling discards. Other scavenging seabirds foraging around the Ebro delta area, such as several species of gulls *Larus* spp. and terns *Sterna* spp., improve breeding performance when this food resource is available (see Oro 1999 and references therein). This area is an important foraging site, not only for locally breeding seabird populations (such as those settled both in the Ebro delta and off the Columbretes Islands; see Oro & Ruiz 1997, Oro 1999), but also for those, such as Balearic shearwaters, breeding in distant colonies (see also Abelló & Oro 1998).

Contribution of breeding success to the population viability of the Balearic shearwater: improving conservation diagnosis

We used a fitness landscape to improve the conservation diagnosis carried out by Oro et al. (2004) of the critically endangered Balearic shearwater by focusing on the influence of breeding success on population viability. Other studies on endangered species of vertebrates (e.g. Cuthbert et al. 2002, Todd et al. 2004) have used different tools of population viability analysis to identify and assess threats and to develop practical management options and decisions (e.g. Norris 2004). In our study, the fitness landscape suggested that current values of breeding success in Balearic shearwaters are normal for the species and that factors potentially affecting this parameter (such as food availability) are not of great conservation concern, at least at the present time and in the study colonies. Nevertheless, incoming fisheries policies are directed at reducing the fishing effort and discard rates (Fluharty 2000), with a consequent reduction of food availability (at least in the short term) for scavenging species. This reduction could, in turn, trigger changes in the seabird community and interspecific relationships (Stenhouse & Montevecchi 1999). For instance, Votier et al. (2004) described how the decrease of discards, particularly when coupled with reduced availability of small pelagic fish, resulted in an increase in predation by the great skua *Stercorarius skua* on other seabirds in the North Sea. However, future research should focus on using direct estimates of prey availability for seabirds rather than indirect proxies, such as fisheries landings.

Our results also suggested that breeding performance needs to be improved sharply to compensate for the current levels of adult mortality in Balearic shearwaters (Oro et al. 2004). Any conservation measure that sets out to enhance breeding success seems difficult to achieve, owing to the practical constraints involved in altering the levels of food availability for shearwaters. In the Mediterranean, fishery activity and the protection of breeding habitats have been identified as key factors for seabird conservation, but management strategies are difficult to apply because of socio-economic constraints (Mínguez et al. 2003). In the case of the Balearic shearwater, conservation strategies have mainly focused on protecting breeding sites (e.g. campaigns to eradicate introduced predators; Ruiz & Martí 2004), but the effectiveness of these measures has not been assessed to date. Future research should concentrate on long-term actions and an effective monitoring programme to maintain the protection of

breeding sites. Nevertheless, factors affecting adult survival, both at sea and in colonies, remain the most important concern for the conservation of the Balearic shearwater. Efforts should focus on reducing this mortality, which seems to occur mostly during dispersal at sea (Arcos & Oro 2004, Oro et al. 2004), both in the western Mediterranean and in the North Atlantic.

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References

- Abad R, Miquel J, Iglesias M, Alvarez F (1998) Acoustic estimation of abundance and distribution of anchovy in the NW Mediterranean. *Scientia Marina* 62:37-43
- Abelló P, Oro D (1998) Offshore distribution and assemblages of breeding seabirds in the Catalano-Balearic Sea (northwestern Mediterranean). *Colonial Waterbirds* 21:422-426
- Agostini VN, Bakun A (2002) 'Ocean triads' in the Mediterranean Sea: physical mechanisms potentially structuring reproductive habitat suitability (with example application to European anchovy, *Engraulis encrasicolus*). *Fisheries Oceanography* 11:129-142
- Arcos JM (2001) Foraging ecology of seabirds at sea: significance of commercial fisheries in the NW Mediterranean. Doctoral dissertation. Universitat de Barcelona. Available: http://tdcat.cesca.es/TESIS_UB/AVAILABLE/TDX-0219102-114337/TOL38.pdf (March 2002)
- Arcos JM, Oro D (2002) Significance of fisheries discards for a threatened Mediterranean seabird, the Balearic shearwater *Puffinus mauretanicus*. *Marine Ecology Progress Series* 239:209-220
- Arcos JM, Oro D (2004) Pardela Balear, *Puffinus mauretanicus*. In: Madroño A, González C, Atienza JC (Eds) Libro Rojo de las Aves de España. Dirección General para la Biodiversidad-SEO/BirdLife, Madrid, pp. 46-50
- Barbraud C, Chastel O (1999) Early body condition and hatching success in the snow petrel *Pagodroma nivea*. *Polar Biology* 21:1-4
- Botsford LW, Castilla JC, Peterson CH (1997) The management of fisheries and marine ecosystems. *Science* 277:509-515
- Carbonell A, Martin P, de Rainieri S, WEDIS team (1998) Discards of the western Mediterranean trawl fleets. *Rapp Comm Int Mer Médit* 35:392-393
- Caswell H (2001) Matrix population models, 2nd edn. Sinauer Press, Sunderland
- Crawley MJ (1993) GLIM for ecologists. Blackwell Scientific, Oxford
- Cuthbert R, Fletcher D, Davis LS (2002) A sensitivity analysis of Hutton's shearwater: prioritizing conservation research and management. *Biological Conservation* 100:163-172
- Fluharty D (2000) Habitat protection, ecological issues, and implementation of the Sustainable Fisheries Act. *Ecological Applications* 10:325-337

- Guisande C, Cabanas JM, Vergara RA, Riveiro I (2001) Effect of climate on recruitment success of Atlantic Iberian sardine *Sardina pilchardus*. *Marine Ecology Progress Series* 223:243-250
- Krebs CJ (2001) *Ecology: the experimental analysis of distribution and abundance*. Benjamin Cummings, San Francisco
- Lewison RL, Crowder LB, Read AJ, Freeman SA (2004) Understanding impacts of fisheries bycatch on marine megafauna. *Trends in Ecology and Evolution* 19:598-604
- Lloret J, Palomera I, Salat J, Sole I (2004) Impact of freshwater input and wind on landings of anchovy (*Engraulis encrasicolus*) and sardine (*Sardina pilchardus*) in shelf waters surrounding the Ebro (Ebro) River delta (north-western Mediterranean). *Fisheries Oceanography* 13:102-110
- Martin JL, Thibault J-C, Bretagnolle V (2000) Black rats, island characteristics, and colonial nesting birds in the Mediterranean: consequences of an ancient introduction. *Conservation Biology* 14:1452-1466
- Mínguez E, Oro D, De Juana E, Martínez-Abraín A (2003) Mediterranean seabird conservation: what can we do? *Scientia Marina* 67S2:3-6
- Montevecchi WA (2002) Interactions between fisheries and seabirds. In: Schreiber EA, Burger J (Eds) *Biology of Marine Birds*. CRC Press, Boca Raton, pp. 27-557
- Myers RA, Worm B (2003) Rapid worldwide depletion of predatory fish communities. *Nature* 423:280-283
- Norris K (2004) Managing threatened species: the ecological toolbox, evolutionary theory and declining-population paradigm. *Journal of Applied Ecology* 41:413-426
- Oro D (1999) Trawler discards: A threat or a resource for opportunistic seabirds? In: Adams NJ, Slotow RH (Eds) *Proceedings of the 22nd International Ornithological Congress, Durban*. Birdlife South Africa, Johannesburg, pp. 717-730
- Oro D, Ruiz X (1997) Seabirds and trawler fisheries in the northwestern Mediterranean: differences between the Ebro Delta and the Balearic Islands areas. *ICES Journal of Marine Science* 54:695-707
- Oro D, Pradel R, Lebreton JD (1999) Food availability and nest predation influence life history traits in Audouin's gull, *Larus audouinii*. *Oecologia* 118:438-445

- Oro D, Aguilar JS, Igual JM, Louzao M (2004) Modelling demography and extinction risk in the endangered Balearic shearwater. *Biological Conservation* 116:93-102
- Pauly D, Alder J, Bennett E, Christensen V, Tyedmers P, Watson R (2003) The future for fisheries. *Science* 302:1359-1361
- Rochet MJ, Trenkel VM (2005) Factors for the variability of discards: assumptions and field evidence. *Canadian Journal of Fisheries and Aquatic Sciences* 62:224-235
- Ruiz A, Martí R (Eds) (2004) *La Pardela Balear*. SEO/Birdlife-Conselleria de Medi Ambient del Govern de les Illes Balears, Madrid
- Saether BE, Engen S (2002) Including uncertainties in population viability analysis using population prediction intervals. In: Beissinger SR, McCullough DR (Eds) *Population viability analysis*. Chicago University Press, IL, pp. 191-212
- Salat J, Garcia MA, Cruzado A, Palanques A, Arín L, Gomis D, Guillén J, de León J, Puigdefàbregas J, Sospedra J, Velásquez ZR (2002) Seasonal changes of water mass structure and shelf-slope exchanges at the Ebro Shelf (NW Mediterranean). *Continental Shelf Research* 22:327-348
- Santolanni A, Arneri E, Barry C, Belardinelli A, Cingolani N, Giannetti G, Kirkwood G (2003) Trends of anchovy (*Engraulis encrasicolus*, L.) biomass in the northern and central Adriatic Sea. *Scientia Marina* 67:327-340
- SAS (2000) *SAS/STAT software: user's guide*. SAS Institute, Cary, NC
- Stenhouse I, Montevecchi WA (1999) Indirect effects of the availability of forage fish and fisheries discards: gull predation on breeding storm-petrels. *Marine Ecology Progress Series* 184:303-307
- Sydeman WJ, Hester MM, Thayer JA, Gress F, Martin P, Buffa J (2001) Climate change, reproductive performance, and diet composition of marine birds in the southern California Current system, 1969-1997. *Progress in Oceanography* 49:309-329
- Tasker ML, Camphuysen CJ, Cooper J, Garthe S, Montevecchi WA, Blaber SJM (2000) The impacts of fishing on marine birds. *ICES Journal of Marine Science* 57:531-547
- Thompson PM, Ollason JC (2001) Lagged effects of ocean climate change on fulmar population dynamics. *Nature* 413:417-420
- Todd CR, Nicol SJ, Koehn JD (2004) Density-dependence uncertainty in population models for the conservation management of trout cod, *Maccullochella macquariensis*. *Ecological Modelling* 171:359-380

Votier SC, Furness RW, Bearhop S, Crane JE, Caldow RWG, Catry P, Ensor K, Hamer KC, Hudson AV, Kalmbach E, Klomp NI, Pfeiffer S, Phillips RA, Prieto I, Thompson DR (2004) Changes in fisheries discard rates and seabird communities. *Nature* 427:727-730

Williams BK, Conroy MJ, Nichols JD (2001) *Analysis and management of animal populations*. Academic Press, San Diego, CA

Chapter 4



Inter-population variation in egg size of an endangered Procellariiform: interplay between body size and feeding ecology

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4.1 Abstract

Inter-population variability in life-history traits can be as large as that among related species. Thus, comparison of traits among populations could identify selective and ecological factors that influence their spatial variability and evolution. However, it is expected that across small geographic scales, nearby populations are constrained by the same ecological features shaping in turn a synchrony in their life-history traits. Here, we studied the spatio-temporal variability in egg size among populations of the critically endangered shearwater *Puffinus mauretanicus*, a top marine predator endemic to the Balearic archipelago (western Mediterranean). We assessed whether this trait

was influenced by maternal body size (as a potential indicator of a genetic component) and feeding ecology (as an indicator of environmental factors, through stable carbon and nitrogen isotope measurements). Strikingly egg size varied among close local populations, an unexpected result at such small spatial scale. Body size differences among those populations (caused by genetic factors) could partially explain such differences. Also, feeding ecology could be playing an important role since $\delta^{15}\text{N}$ values suggested inter-population differences in trophic position showing a similar general pattern to egg size, suggesting a nutritional link between them (i.e. egg size affected by differences in feeding resources and/or behaviour). Isotopic $\delta^{13}\text{C}$ values suggested that breeders from different local populations showed similar foraging habits with respect to benthic- vs. pelagic-based food-webs. In contrast, egg size did not vary among years as did breeding performance, suggesting that a differential temporal window could affect both breeding parameters in relation to food availability. The absence of relationship between breeding performance and egg size suggested that larger eggs may only confer an advantage in harsh, low-quality conditions, while alternatively parental quality could greatly affect breeding performance. Results showed that inter-population differences in life-history traits may occur at small spatial scales, influenced by genetic and environmental factors.

4.2 Introduction

Inter-population variability in life-history traits largely has been ignored but it has received more attention in recent years (Williams et al. 2003, Frederiksen et al. 2005, Harris et al. 2005). Variation of life-history traits among populations can be as large as that among related species, and comparison among populations can be a powerful tool to identify selective factors that influence their spatial variability and evolution (Barbraud et al. 1999, Dhondt 2001, Weimerskirch 2002). Although it has been suggested that correlations between traits among different populations might be exceptions rather than the general rule (Weimerskirch 2002), spatial synchrony recorded in population dynamics suggests that similar environmental conditions could cause synchrony in life-history traits (Tedesco et al. 2004 and references therein). Most studies examining patterns and processes in inter-population variation have been done over large spatial scales, and these variations are usually responses to variation in local environmental conditions (Frederiksen et al. 2005, Harris et al. 2005, Ruttenberg et al.

2005). However, across small geographic scales, comparisons are constrained to nearby populations (c.f. Ruttenberg et al. 2005), and life-history traits should be similar owing to the constraints imposed by the same ecological features.

In the context of life-history theory, egg size has been widely studied because of its high variability (Hipfner 2000), which in turn often varies widely within species, and is a representative trait of maternal breeding effort (Bernardo 1996). In the case of seabirds, egg size can also reflect environmental stochastic factors such as weather, oceanographic conditions, and marine resource availability around the colony during breeding (Oro 1996, Ruiz et al. 2000, Michel et al. 2003). Within seabirds, an ideal group to study potential factors affecting egg size is the Procellariiformes. They lay a single egg (Warham 1990), and females can adjust their reproductive investment by breeding intermittently or by altering size or composition of eggs (Weidinger 1996). However, since Procellariiformes have the largest eggs in relation to their body size, within seabirds (Whittow 2002), this could constrain their ability to adjust the size of this life-history trait, since below and over this relationship, eggs would not be viable and possible respectively. In addition, egg size may increase until a threshold value with parental quality (e.g. age and breeding experience, see Richdale 1949, Brooke 1990, Michel et al. 2003, Sagar et al. 2005) or with parental body condition at the time of breeding (Bolton 1991, Barbraud & Chastel 1999). When egg size increases with maternal body size (Michel et al. 2003), a genetic component may be involved in egg size determination, although this relationship is not always clear within birds in general (Christians 2002) and Procellariiformes in particular (Mougin 1998). Finally, environmental and parental factors are not mutually exclusive and effects of environmental stochasticity on breeding performance may depend on the quality of the parents (Saether et al. 1997). Environmental variability could promote fitness consequences such as quality of offspring: for instance lower mass at fledging would reduce the probability of survival until breeding (Weimerskirch et al. 2001). Furthermore, much research has been devoted to study the consequences of egg size on fitness components, although results are controversial. Several studies found that increased egg size may lead to increase hatchling mass (Hipfner 2000), chick size in the first week after hatching (Christians 2000), rate of post-hatching development (Hipfner 2000, Quillfeldt & Peter 2000), and consequently reproductive success (Michel et al. 2003). However, the absence of an egg size effect on breeding performance has been also reported in birds (Meathrel et al. 1993, Christians 2000).

Here, we studied the inter-population variability of egg size of the critically endangered Balearic shearwater *Puffinus mauretanicus* in different local populations (i.e. breeding colonies). The species is endemic to the Balearic archipelago, with a small breeding population (less than 2000 pairs and maximum distance between colonies of ~ 300 km) (Ruiz & Martí 2004). We assessed (1) the spatio-temporal variability of egg size in a relatively small geographical area, (2) maternal body size as potential indicator of a genetic component in egg size determination, (3) differences in feeding ecology among local populations as potential factor influencing egg size variability and (4) the potential effect of egg size on breeding performance, as measured by hatching and breeding success. We predicted that local populations, affected similarly by predominant ecological features at such small spatial scale, should in turn show synchrony in egg size variability. This should be especially true at the local populations under study, with common physical features at breeding sites (see Louzao et al. 2006a) being potential determinants of variation in life-history traits at small spatial scale (e.g. Birkhead 1977, Parrish 1995, Barbosa et al. 1997, Genovart et al. 2003b). Furthermore, we predicted: as egg size depends at least partially on feeding resources, we did not expect differences among local populations, owing that most feeding birds concentrate on very few foraging grounds far from the breeding sites (Ruiz & Martí 2004, Louzao et al. 2006b). For such purpose stable isotope analyses of nitrogen ($^{15}\text{N}/^{14}\text{N}$, expressed as $\delta^{15}\text{N}$) and carbon ($^{13}\text{C}/^{12}\text{C}$, expressed as $\delta^{13}\text{C}$) were performed and compared among local populations. In seabirds, stable isotopes are a highly valuable tool to discriminate between potential food resources, as indicated by differential trophic levels, and feeding habits (Hobson et al. 1994, Forero & Hobson 2003).

4.3 Material and methods

Species details and study area

The Balearic shearwater is a medium-sized (ca. 500g), and slightly dimorphic Procellariiform, with males about 5% larger than females (Genovart et al. 2003a). Birds start to breed at 3 years-old (Oro et al. 2004). Females lay a single egg that both parents incubate during 48-52 days, and the chick-rearing period extends ca. 65-70 days (Oro et al. 2004, Ruiz & Martí 2004). The species is specialised at foraging on shoals of small pelagic fishes, although a bioenergetics model estimated that over 40% of the energetic

demands of the total population was met by trawling discards (Arcos & Oro 2002). However, Louzao et al. (2006b) hypothesized that discards represent a quantitatively important, yet secondary food resource, taken opportunistically in the absence of alternative prey (cf. Arcos et al. 2006). Shearwaters are present in shallow shelf and coastal waters characterised by frontal systems in areas close to the breeding colonies on the Iberian continental shelf, but other areas within the foraging range of the species could be important foraging areas such as the Gulf of Lions or Algerian coast (North Africa) (Ruiz & Martí 2004, Louzao et al. 2006b). Adult survival is the most highly sensitive demographic parameter in the population dynamics of the species and consequently the most important for its conservation (Oro et al. 2004, Louzao et al. 2006a).

Fieldwork was conducted at seven breeding colonies from Mallorca, Menorca and Eivissa islands (4, 1 and 2 colonies, respectively) (Figure 4.1). However, we only analysed egg size from three colonies: Sa Cella (39° 36' N, 2° 22'; noted by CEL), and Conills (39° 30' N, 2° 28' E; noted by CON) in Mallorca, and Mola de Maó (39° 53' N, 4° 19' E; noted by MOL) in Menorca through the 2001-2004 breeding seasons, and excluded other colonies (Sa Dragonera, Malgrats, Conillera and Es Bosc) because data were too sparse (see Table 4.1). In the three colonies of study, breeding population ranged from ca. 45 pairs (CON) to ca. 200 pairs (CEL and MOL). Colonies were located in deep caves (CEL), between alluvia in coastal cliffs (MOL) and islets (CON), and only MOL was accessible by land.

Field procedures: egg measures, breeding performance, and adult sampling

Colonies were visited through the incubation period (from late February to late April) to monitor egg laying and take egg measurements. All accessible active nests (i.e. nests where a yearly egg was found) were marked and when possible, we measured the total egg length and breadth to the nearest 0.01 mm with a digital calliper. Egg size was estimated as an egg volume index (in cm³) = 0.00051 × egg length × (egg breadth)² (Hoyt 1979). Visits were repeated in mid May and June to record hatchlings and fledglings, respectively, at each monitored nest. Hatching and breeding success (as the percentage of hatchlings and fledglings by eggs laid, respectively) were estimated only in colonies from Mallorca, since MOL was only visited during the incubation period.

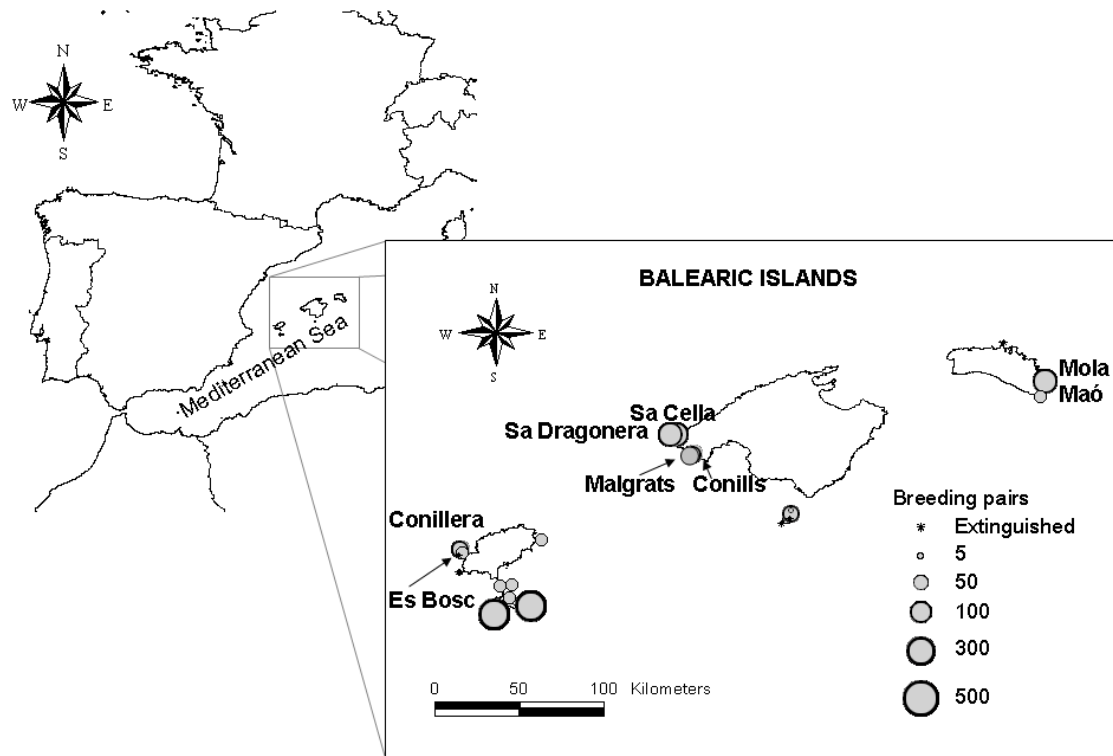


Figure 4.1 Study area and location of the breeding colonies of the Balearic shearwater. Cited colonies in the text are located.

We captured adult breeders (identified when incubating or attending a young chick in the nest) and took morphological measurements (head plus bill length, bill depth at nostrils, and tarsus with a digital calliper to the nearest 0.01 mm). When possible, we extracted ~1 mL of blood, which was preserved in 70% ethanol at room temperature and further used for both molecular sexing and stable isotope analysis (see Genovart et al. 2003b and Forero et al. 2002a respectively, for methodological details in samples processing). All individuals from Menorca were molecularly sexed although this was not possible for all adults from Mallorca, for which a discriminant function was applied (Genovart et al. 2003a).

Feeding ecology: stable isotope analyses

Ethanol was removed from blood samples prior to stable isotope analysis by decanting and then successive rinsing in distilled water followed by freeze-drying. Stable isotope assays were performed on 1 mg sub-samples of homogenised materials by loading into tin cups and combusting at 1200° C in a Robo-Prep elemental analyser

the released CO₂ and N₂ were analysed using a continuous-flow isotope ratio mass spectrometer (CFIRMS) with every 5 unknowns separated by 2 laboratory standards (albumen). The isotopic composition of the samples was expressed using δ -notation or deviation from international standards in parts per thousand (‰) using the formula: $\delta X = [(R_{\text{sample}} - R_{\text{standard}}) / R_{\text{standard}}] \times 1000$ where X is ¹³C or ¹⁵N and R is the proportion ¹³C / ¹²C or ¹⁵N / ¹⁴N. The reference standard for ¹³C was the PeeDee Belemnite (PDB) limestone and for ¹⁵N atmospheric N₂. The analytical precision of the procedure was estimated at ± 0.1 and ± 0.3 ‰ for stable-carbon and nitrogen isotope analyses, respectively.

We only gathered sufficient information on both isotope signatures to compare feeding ecology among colonies at the local population level. For seabirds of the size of the Balearic Shearwater, stable isotope signatures of whole blood integrate diet information over a period of at least four weeks (Hobson & Clark 1992). Blood samples from breeders were obtained only during 2003. Because stable isotopes signature could vary depending on the period of the breeding cycle (c.f. Schmutz & Hobson 1998), we took blood samples of adult breeders during late incubation and early hatching period, representing the diet of shearwaters during the peak incubation period (from mid March to mid April).

Statistical analysis

We used Generalized Linear Mixed Models (GLMM MIXED procedure, in SAS Institute 2000) to assess spatio-temporal variability in egg size among local populations, and also the effect of egg size on breeding performance. Some egg volumes (measured in different years) belonged to the same nest and its identity was introduced in GLMMs as random to account for pseudoreplication (Littell et al. 1986). We assumed that the probability of change of breeding pairs between nests was very low due to the extremely low breeding dispersal of Procellariiformes (Warham 1990). We assembled the best-fit model iteratively using a backwards stepwise procedure starting from a saturated model, and rejecting the variables with the lowest explanatory power one at a time (variables were discarded whenever $\alpha > 0.05$ and retained otherwise). However, when analysing inter-population differences on maternal body size and feeding ecology ($\delta^{15}\text{N}$ and $\delta^{13}\text{C}$) we did not introduce random effects since comparisons were performed at the local population but not individual

level, in different females and blood samples only in 2003, respectively. Therefore, Generalized Linear Models (GLMs GENMOD procedure, in SAS Institute 2000) were used instead. Significance level was set at 0.05 and when appropriate contrast analyses were performed for both GLMMs and GLMs.

Firstly, we performed a preliminary analysis to test if egg volume could depend, at least partially, on maternal body size. Therefore, we needed to calculate a body size index (BSI hereafter) for reducing body morphometrics to a unique variable (e.g. Forero et al. 2002b, Genovart et al. 2003a). We used principal component analysis (PCA in SPSS 12.0) of the three morphometrics, particularly the first axis (PC1) since it can summarise most of the variability of those body measurements (Legendre & Legendre 1998), to combine linear body measurements of breeding females. We extracted the BSI from only one factor, PC1, and accounted for 51% of the total variance. For this preliminary analysis, the saturated model was designed with colony, female BSI, and their interaction as fixed effects ($v = \text{BSI} + c + \text{BSI} * c$), and we fitted female identity, instead of nest identity since we tested only female effects in this analysis, and year as random terms. No significant effect of maternal body size on egg size was found (see Results), so further analyses did not include this factor as explanatory variable.

After this preliminary analysis, we assessed the spatio-temporal variability in egg size and the saturated model was designed with colony, year, and their interaction as fixed effects ($v = c + t + c * t$), and nest identity as random effect. Similarly, for breeding performance (both hatching and breeding success), the saturated model was designed as the additive model of egg size, colony and year effect (breeding performance = $v + c + t$), and nest identity as random. No interaction terms were introduced as fixed effects due to the small sample size. When analysing inter-population differences on maternal body size and feeding ecology only colony was introduced as fixed effect.

Colony (denoted by c), year (denoted by t), and both female and nest identity were treated as categorical variables and egg size (denoted by v), BSI, and stable isotopes as continuous variables. When modelling egg size, maternal body size, and stable isotope data were fitted using an identity-link function. Hatching and breeding success were treated as binomial dependent variables and fitted using a logit-link function.

Table 4.1 Descriptive statistics of egg volume (mean \pm SE in cm³, sample sizes in parentheses) of Balearic shearwaters in all surveyed colonies through the study period (2001-2004). Some data have been introduced as informative but were not analysed because of their small sample size.

Main island	Colony	Year			
		2001	2002	2003	2004
Mallorca	Sa Cella	56.81 \pm 0.53 (74)	57.27 \pm 0.51 (79)	57.7 \pm 0.52 (69)	56.38 \pm 0.68 (45)
	Dragonera				62.55 (1)
	Conills	55.37 \pm 0.98 (12)	55.36 \pm 1.37 (17)	55.84 \pm 1.03 (11)	53.48 \pm 2.13 (5)
	Malgrats	58.64 \pm 1.07 (17)			
Menorca	Mola de Maó		51.99 \pm 1.22 (13)	53.19 \pm 1.36 (11)	53.25 \pm 1.21 (13)
Eivissa	Conillera			58.07 \pm 2.85 (3)	
	Es Bosc				57.41 (1)

4.4 Results

We collected information on 349 eggs from three colonies of Balearic shearwaters ($n = 265, 45$ and 37 at CEL, CON and MOL respectively) (Figure 4.1). Considering all the study years and the colonies with sufficient sample sizes, mean (\pm SE) egg size (in cm³) ranged from 51.99 ± 1.22 to 58.64 ± 1.07 (Table 4.1). We assessed hatching and breeding success from 263 and 259 breeding attempts (measured eggs), respectively. Hatching and breeding success ranged between $0.75 - 1$ and $0.68 - 1$ respectively, in both colonies from Mallorca. In order to minimize manipulation and due to logistic constraints, we did not capture nor measure all breeding pairs. We only gathered information on body morphometrics of 70 breeding females ($n = 51, 9$ and 10 at CEL, CON and MOL respectively). Moreover, during the peak incubation period in 2003, we took 47 blood samples of adults ($n = 23, 9$ and 15 at CEL, CON and MOL respectively) for stable isotope analyses.

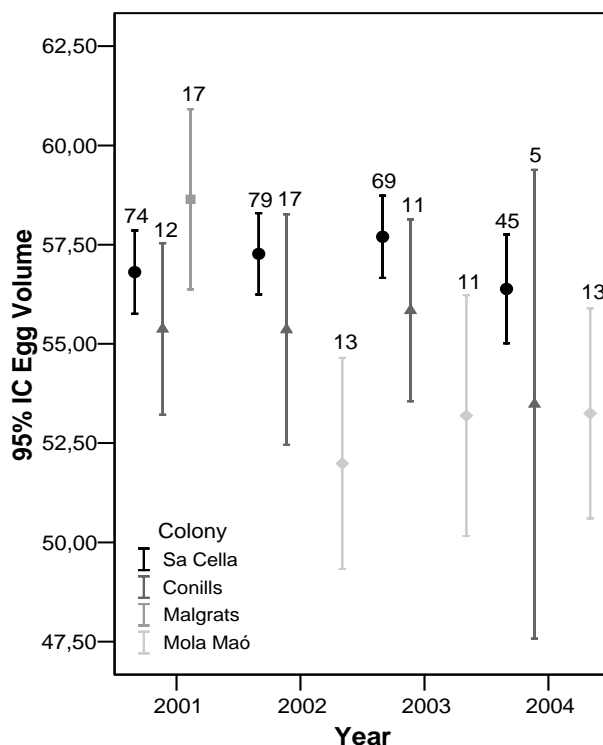


Figure 4.2 Mean egg volume and 95% confidence interval of the analysed four colonies during the study period (2001-2004). Sample sizes are shown. Mola de Maó was not sampled in 2001 whereas Malgrats was only sampled in 2001.

Spatio-temporal patterns of egg size and its influence on breeding performance

In relation to egg size and female body size relationship, we gathered information on BSI of 66 breeding females associated with 119 egg measurements ($n = 48, 8$ and 10 females at CEL, CON and MOL, respectively). From the saturated model ($v = \text{BSI} + c + \text{BSI} * c$) the backward procedure finally selected the colony effect ($F_2 = 6.41, P = 0.003$) as significant. Therefore, since we did not find any significant relationship between egg size and female BSI, in a sub-sample of 119 eggs, we were able to recover the information gathered on the total of 349 eggs to analyse its spatio-temporal variability. Again, from the saturated model ($v = c + t + c * t$) the colony effect was only significant ($F_2 = 13.38, P < 0.001$). Contrast analyses showed that differences between CEL vs. MOL and CON vs. MOL were significant ($F_1 = 26.0, P < 0.001$ and $F_1 = 6.97, P = 0.01$), whereas differences between CEL vs. CON were not ($F_1 = 2.60, P = 0.109$) (Figure 4.2).

For both hatching and breeding success (saturated model: breeding performance = $v + c + t$) the final model found only the year effect as significant for both breeding parameters (hatching success $F_3 = 3.42, P = 0.020$; breeding success $F_3 = 3.01, P = 0.033$).

Inter-population comparison of maternal body size and feeding ecology

Regarding maternal body size comparison at the population level, we found a significant colony effect in BSI ($\chi^2_2 = 26.42$, $P < 0.001$). Contrast analyses showed that only differences between CEL vs. MOL ($\chi^2_1 = 26.39$, $P < 0.001$), and CON vs. MOL ($\chi^2_1 = 12.62$, $P < 0.001$) were significant. Females from MOL were significantly smaller than those from CEL and CON (Figure 4.3).

We found a significant colony effect in $\delta^{15}\text{N}$ values ($\chi^2_2 = 31.51$, $P < 0.001$), but not in $\delta^{13}\text{C}$ values ($\chi^2_2 = 0.57$, $P = 0.753$). Values of mean $\delta^{15}\text{N}$ ranged from $9.3 \pm 0.7\text{‰}$ in MOL to $10.4 \pm 0.3\text{‰}$ in CEL, CON presenting intermediate values. All possible contrast analyses of $\delta^{15}\text{N}$ were significant (CEL vs. CON: $\chi^2_1 = 5.22$, $P = 0.022$; CEL vs. MOL: $\chi^2_1 = 31.51$, $P < 0.001$; and CON vs. MOL: $\chi^2_1 = 8.65$, $P = 0.003$) indicating possible trophic segregation (i.e. foraging at different trophic position) between colonies, with individuals from MOL foraging at lower trophic levels than in CEL and CON. On the other hand, mean $\delta^{13}\text{C}$ values ranged from $-19.4 \pm 0.3\text{‰}$ in CON to $-19.3 \pm 0.2\text{‰}$ in MOL (see Figure 4.4), suggesting similar feeding habits among populations.

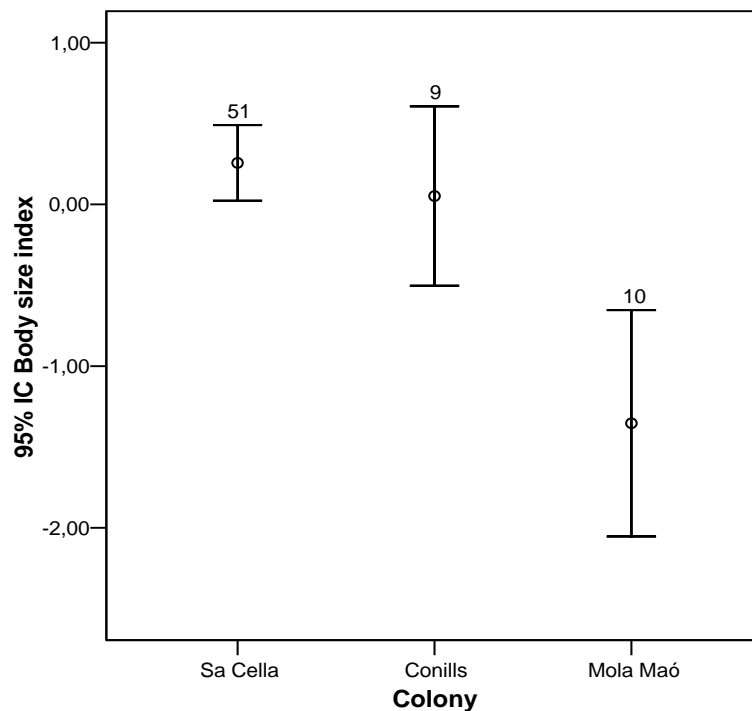


Figure 4.3 Body size index of 70 breeding females of the Balearic shearwater, in the three colonies during 2001-2004. Sample sizes are shown for each colony.

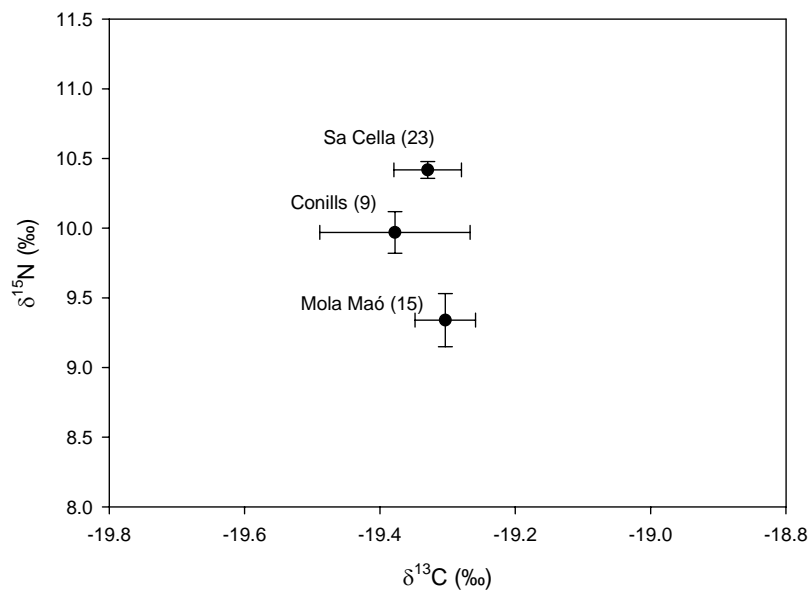


Figure 4.4 Mean (\pm SE) stable carbon and nitrogen isotope ratios in blood samples from adults representing the diets of Balearic shearwaters during the peak incubation period (from mid March to mid April) in the three studied colonies in 2003. Sample sizes are shown.

4.5 Discussion

Inter-population variation of life-history traits have been often recorded not only in birds (Christians 2000, Frederiksen et al. 2005, Harris et al. 2005), but also on other vertebrates such as reptiles and fishes (Sinervo 1990, Duponchelle et al. 2000, Ruttenberg et al. 2005). Factors affecting egg size and its variability have been widely studied, together with its potential association with breeding performance (Croxall et al. 1992, Meathrel et al. 1993, Barbraud & Chastel 1999, Hipfner 2000). These studies yielded divergent conclusions about the variability of egg size among close species, and even within the same species among local populations (e.g. Procellariiformes, Amundsen 1995, Mougín 1998 and references therein). Nevertheless, differences in life-history traits have been recorded between distant populations (and thus attributed to differential environments) or in close populations with distinct ecological features (physical breeding habitat, density of predators).

Spatial patterns of egg size variability

Strikingly our results showed variability of egg size among neighbouring local populations, even though we did not expect differences at such small spatial scale, with study colonies sharing the same ecological features (e.g. breeding habitat, oceanographic conditions), and thus hindering local adaptations and potential variability in life-history traits. The variability found in our study could partly be explained by unexpected differences in feeding ecology among colonies: mean $\delta^{15}\text{N}$ values revealed evidence of inter-population variability in trophic position, while $\delta^{13}\text{C}$ values suggested that breeders from different local populations showed similar feeding habits related to both benthic- vs. pelagic-based food-webs (i.e. fishery discards and small pelagic prey, respectively see Arcos & Oro 2002, Louzao et al. 2006a). However, all of our interpretations from the stable isotopes need to be taken with caution. We did not investigate in detail the isotopic nature of the food-webs used by each of the populations, and the baseline $\delta^{15}\text{N}$ values may differ causing different $\delta^{15}\text{N}$ values for food items at the same trophic level (Schell et al. 1998). However, this seemed unlikely due to the proximity of the colonies (CEL and CON separated ~ 15 km and Mallorca and Menorca ~ 160 km), and to the fact that we only observed a difference in $\delta^{15}\text{N}$ values and not in $\delta^{13}\text{C}$ values among colonies. Differences in feeding ecology among populations have been well documented in Procellariiformes (Montevecchi et al. 1992 and references therein, Forero et al. 2005, Peck & Congdon 2005), but always at larger spatial scale than that of our study and attributed to contrasting local environmental conditions affecting each local population (see also Frederiksen et al. 2005).

Differences among colonies in both trophic position and egg size showed the same general pattern (larger egg size potentially related to higher $\delta^{15}\text{N}$ mean values), suggesting a nutritional link between them (i.e. egg size affected by differences in feeding resources and/or behavioural plasticity). Both trophic position and in turn egg size could probably be influenced by parental body size, a potential indicator of a genetic component, since it could probably mediate prey selection (i.e. larger body sizes foraging upon higher trophic levels) (cf. Barbraud et al. 1999, Forero et al. 2002b) and conditioned egg size. This genetic hypothesis would presumably explain differences between colonies from Mallorca and Menorca, actually supported by a genetic introgression of its smaller sibling species, the Yelkouan shearwater *P. yelkouan* in the latter colony (Genovart et al. *in press*). Thus, smaller individuals from Menorca

could lay smaller eggs than breeders from Mallorca. Similarly, Amundsen (1995) and Mougin (1998) found differential egg size due to local genetic differences in different subspecies of Snow petrels *Pagodroma nivea* and Cory's shearwater *Calonectris diomedea* respectively, with larger individuals laying bigger eggs. Yet, we surprisingly found significant differences in egg size between Conills and Malgrats ($F_{1,29} = 4.62$, $P = 0.041$) (see Figure 4.1, 4.2 and Table 4.1), two local populations located in two distinct islets separated by only 100 m. This result supported the evidence that nearby populations of Balearic shearwaters with similar genetic component and constrained by the same ecological features, could show asynchrony at least in a single life-history trait. This variability could generate in turn differences in vital rates between close populations, avoiding the synchronization of population fluctuations, an issue that has recently received attention due to its importance in local population extinction and the consequences for wildlife conservation (e.g. Earn et al. 2000).

Temporal variability in egg size and its effect on breeding performance

We found no significant variation in egg size among years, in agreement with data available for single-egg systems (i.e., Procellariiformes: Brooke 1990, Mougin 1998, Sagar et al. 2005, but see Croxall et al. 1992, Weidinger 1996, Michel et al. 2003). Nevertheless, temporal variability affected the breeding performance of Balearic shearwaters, presumably related to temporal stochasticity of prey density (Louzao et al. 2006a). Egg size could be a more conservative evolutionary trait regarding food availability than breeding performance, probably due to the differential temporal window affecting both breeding parameters: egg formation represents a short-term energetic investment compared to the long-term period affecting breeding performance (see Ricklefs 1990). The absence of any relationship between egg size and breeding performance found in our study also agreed with other studies on Procellariiforms recording no relationship between these two parameters (Meathrel et al. 1993, Weidinger 1996, Mougin 1998, but see Croxall 1992). Christians (2002) suggested that larger eggs may only confer an advantage in harsh, low-quality conditions, when females invest part of their energetic reserves laying an egg of sufficient quality (i.e. minimum or similar egg size) to produce offspring of high quality (Ricklefs 1990, Weimerkirch et al. 2001). Alternatively, different authors have pointed out the importance of parental quality rather than egg size in offspring

survival (Bolton 1991, Saether et al. 1997). However, factors related to maternal characteristics (i.e. female age, mass and size), and foraging skills (e.g., food availability and quality) are difficult to disentangle since they are not independent (Christians 2000).

In summary, inter-population differences in genetic and ecological factors could interact to determine egg size variability in the Balearic shearwater at small spatial scales. We showed that differences in feeding ecology could be one of the consequences of such number of factors, shaping the parallel trend between egg size and trophic position among local populations. Further research is needed to explore the variability of life-history traits in animals, bridging the gaps between ecological and genetic determinants at different spatio-temporal scales. Ultimately, differences in life-history traits in close populations could be an evolutionary mechanism to avoid coherent oscillations in population dynamics and their extinction.

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References

- Amundsen T (1995) Egg size and early nestling growth in the snow petrel. *The Condor* 97:345-351
- Arcos JM, Oro D (2002) Significance of fisheries discards for a threatened Mediterranean seabird, the Balearic shearwater *Puffinus mauretanicus*. *Marine Ecology Progress Series* 239:209-220
- Arcos JM, Louzao M, Oro D (2006) Fishery Ecosystem Impacts and Management in the Mediterranean: Seabirds Point of View. In: Nielsen J, Dodson J, Friedland K, Hamon T, Hughes N, Musick J, Verspoor E (Eds) *Proceedings of the Fourth World Fisheries Congress: Reconciling Fisheries with Conservation*. American Fisheries Society, Symposium 49, Bethesda, Maryland, in press.
- Barbosa A, Moreno J, Potti J, Merino S (1997) Breeding group size, nest position and breeding success in the chinstrap penguin. *Polar Biology* 18:410-414
- Barbraud C, Chastel O (1999) Early body condition and hatching success in the snow petrel *Pagodroma nivea*. *Polar Biology* 21:1-4
- Barbraud C, Weimerskirch H, Robertson GG, Jouventin P (1999) Size related life-history traits: insights from a study of snow petrels (*Pagodroma nivea*). *Journal of Animal Ecology* 68:1179-1192
- Bernardo J (1996) The particular maternal effect of propagule size, especially egg size: patterns, models, quality of evidence and interpretations. *American Zoologist* 36:216-236
- Birkhead TR (1977) The effect of habitat and density on breeding success in the common guillemot (*Uria aalge*). *Journal of Animal Ecology* 46:751-764
- Bolton M (1991) Determinants of chick survival in the lesser black-backed gull: relative contributions of egg size and parental quality. *Journal of Animal Ecology* 60:949-960
- Brooke ML (1990) *The Manx shearwater*. London: Poyser
- Christians JK (2002) Avian egg size: variation within species and inflexibility within individuals. *Biological Reviews* 77:1-26
- Croxall JP, Rothery P, Crisp A (1992) The effect of maternal age and experience on egg-size and hatching success in wandering albatrosses *Diomedea exulans*. *Ibis* 134: 219-228

- Dhondt AA (2001) Tradeoffs between reproduction and survival in tits. *Ardea* 89 (special issue):155-166
- Duponchelle F, Cecchi P, Corbin D, Nuñez J, Legendre M (2000) Variations in fecundity and egg size of female Nile Tilapia, *Oreochromis niloticus*, from man-made lakes of Côte D'Ivoire. *Environmental Biology of Fishes* 57:155-170
- Earn DJD, Levin SA, Rohani P (2000) Coherence and conservation. *Science* 290:1360-1364
- Forero MG, Tella JL, Hobson KA, Bertellotti M, Blanco G (2002a) Conspecific food competition explains variability in colony size: a test in Magellanic Penguins. *Ecology* 83:3466-3475
- Forero MG, Hobson KA, Bortolotti GR, Donázar JA, Bertellotti M, Blanco G (2002b) Food resource utilisation by the Magellanic penguin evaluated through stable-isotopes analysis: segregation by sex and age and influence on offspring quality. *Marine Ecology Progress Series* 234:289-299
- Forero MG, Hobson KA (2003) Using stable isotopes of nitrogen and carbon to study seabird ecology: applications in the Mediterranean seabirds community. *Scientia Marina* 67S2:23-32
- Forero MG, González-Solís J, Hobson KA, Donázar JA, Bertellotti M, Blanco G, Bortolotti GR (2005) Stable isotopes reveal trophic segregation by sex and age in the southern giant petrel in two different food webs. *Marine Ecology Progress Series* 296:107-113
- Frederiksen M, Harris MP, Wanless S (2005) Inter-population variation in demographic parameters: a neglected subject? *Oikos* 111:209-214
- Genovart M, McMinn M, Bowler D (2003a) A discriminant function for predicting sex in the Balearic Shearwater. *Waterbirds* 26:72-76
- Genovart M, Jover L, Ruiz X, Oro D (2003b) Offspring sex ratios in subcolonies of Audouin's gull, *Larus audouinii*, with differential breeding performance. *Canadian Journal of Zoology* 81:905-910
- Genovart M, Oro D, Juste J, Bertorelle G (*in press*) What genetics tell us about the conservation of the critically endangered Balearic Shearwater? *Biological Conservation*
- Harris MP, Anker-Nilssen T, McCleery RH, Erikstad KE, Shaw DN, Grosbois V (2005) Effect of wintering area and climate on the survival of adult Atlantic puffins

- Fratercula arctica* in the eastern Atlantic. Marine Ecology Progress Series 297:283-296
- Hipfner JM (2000) The effect of egg size on post-hatching development in the Razorbill: an experimental study. Journal of Avian Biology 31:112-118
- Hobson KA, Clark RG (1992) Assessing avian diets using stable isotopes II: factors influencing diet-tissue fractionation. The Condor 94:189-197
- Hobson KA, Piatt JF, Pitocchelli J (1994) Using stable isotopes to determine seabird trophic relationships. Journal of Animal Ecology 63:786-798
- Hoyt DF (1979) Practical methods of estimating volume and fresh weight of birds eggs. The Auk 96:73-77
- Legendre P, Legendre L (1998) Numerical ecology. Amsterdam: Elsevier
- Littell RC, Milliken GA, Stroup WW, Wolfinger RS (1996) SAS system for mixed models. Cary, SAS Institute, Inc
- Louzao M, Igual JM, McMinn M, Aguilar JS, Triay R, Oro D (2006a) Small pelagic fish, trawling discards and breeding performance of the critically endangered Balearic Shearwater: improving conservation diagnosis. Marine Ecology Progress Series 318:247-254
- Louzao M, Hyrenbach KD, Arcos JM, Abelló P, Gil de Sola L, Oro D (2006b) Oceanographic habitat of an endangered Mediterranean Procellariiform: implications for Marine Protected Areas. Ecological Applications (in press)
- Martin T E (1987) Food as a limit on breeding birds: a life-history perspective. Annual Review of Ecology and Systematics 18:453-487
- Meathrel CE, Bradley JS, Wooller RD, Skira IJ (1993) The effect of parental condition on egg-size and reproductive success in short-tailed shearwaters *Puffinus tenuirostris*. Oecologia 93:162-164
- Michel P, Ollason JC, Grosbois V, Thompson PM (2003) The influence of body size, breeding experience and environmental variability on egg size in the northern fulmar (*Fulmarus glacialis*). Journal of Zoology 261:427-432
- Montevecchi WA, Birt-Friesen VL, Cairns DK (1992) Reproductive energetics and prey harvest of Leach's storm-petrels in the Northwest Atlantic. Ecology 73:823-832
- Mougin J-L (1998) Factors affecting egg dimensions and breeding success in the Cory's shearwater (*Calonectris diomedea*) of Selvagem Grande. Journal für Ornithologie 139:179-184

- Oro D (1996) The effects of trawler discard availability on the egg-laying and the breeding success of the Lesser Black-backed Gull *Larus fuscus* in western Mediterranean. Marine Ecology Progress Series 132:43-46
- Oro D, Aguilar JS, Igual JM, Louzao M (2004) Modelling demography and extinction risk in the endangered Balearic shearwater. Biological Conservation 116:93-102
- Parrish JK (1995) Influence of group size and habitat type on reproductive success in Common Murres (*Uria aalge*). The Auk 112:390-401
- Peck DR, Congdon BC (2005) Colony-specific foraging behaviour and co-ordinated divergence of chick development in the wedge-tailed shearwater *Puffinus pacificus*. Marine Ecology Progress Series 299:289-296
- Quillfeldt P, Peter H-U (2000) Provisioning and growth in chicks of Wilson's storm-petrels (*Oceanites oceanicus*) on King George Island, South Shetland Islands. Polar Biology 23:817-824
- Richdale LE (1949) Buller's mollymawk: incubation data. Bird-banding 20:127-141
- Ricklefs RE (1990) Seabird life histories and the marine environment: some speculations. Colonial Waterbirds 13:1-6
- Ruiz X, Jover L, Pedrocchi V, Oro D, González-Solís J (2000) How costly is clutch formation in Audouin's gull, *Larus audouinii*? Journal of Avian Biology 31:567-575
- Ruiz A, Martí R (Eds) (2004) La Pardela Balear. Madrid: SEO/BirdLife-Conselleria de Medi Ambient (Govern de les Illes Balears)
- Ruttenberg BI, Haupt AJ, Chiriboga AI, Warner RR (2005) Patterns, causes and consequences of regional variation in the ecology and life history of a reef fish. Oecologia 145:394-403
- Sinervo B (1990) The evolution of maternal investment in lizards: an experimental and comparative analysis of egg size and its effect on offspring performance. Evolution 44:279-294
- Saether B-N, Lorentsen S-H, Tveraa T, Andersen R, Pedersen HC (1997) Size-dependent variation in reproductive success of a long-lived seabird, the Antarctic Petrel (*Thalassoica antarctica*). The Auk 114:333-340
- Sagar PM, Unwin MJ, Stahl JC, Warham J (2005) Variation in the egg size of Buller's albatross (*Thalassarche bulleri*) eggs. New Zealand Journal of Zoology 32:171-180
- Schell DM, Barnett BA, Vinette K (1998) Carbon and nitrogen isotope ratios in zooplankton of the Bering, Chukchi and Beaufort Seas. Marine Ecology Progress Series 162:11-23

- Schmutz JA, Hobson KA (1998) Geographic, temporal, and age-specific variation in diets of Glaucous gulls in western Alaska. *The Condor* 100:119-130
- Serventy DL, Curry PJ (1984) Observations on colony size, breeding success, recruitment and inter-colony dispersal in a Tasmanian colony of Short-tailed Shearwaters *Puffinus tenuirostris* over a 30-year period. *Emu* 84:71-79
- Tedesco PA, Hugueny B, Paugy D, Fermon Y (2004) Spatial synchrony in population dynamics of West African fishes: a demonstration of an intraspecific and interspecific Moran effect. *Journal of Animal Ecology* 73:693-705
- Thompson DR, Furness RW, Lewis SA (1995) Diets and long-term changes in $\delta^{15}\text{N}$ and $\delta^{13}\text{C}$ values in Northern Fulmars *Fulmarus glacialis* from two northeast Atlantic colonies. *Marine Ecology Progress Series* 125:3-11
- Warham J (1990) *The petrels: their ecology and breeding systems*. London Academic Press, London
- Weimerskirch H (2002) Seabird demography and its relationship with the marine environment. In: Schreiber EA, Burger J (Eds) *Biology of Marine Birds*. CRC Press, Boca Raton, pp. 115-135
- Weimerskirch H, Zimmermann L, Prince PA (2001) Influence of environmental variability on breeding effort in a long-lived seabird, the yellow-nosed albatross. *Behavioral Ecology* 12:22-30
- Weidinger K (1996) Egg variability and hatching success in the Cape Petrel *Daption capense* at Nelson Island, South Shetland Islands, Antarctica. *Journal of Zoology* 239:755-758
- Whittow GC (2002) Seabird reproductive physiology and energetics. In: Schreiber EA, Burger J (Eds) *Biology of Marine Birds*. CRC Press, Boca Raton, pp. 409-437
- Williams CK, Ives AR, Applegate RD (2003) Population dynamics across geographical ranges: time-series analyses of three small game species. *Ecology* 84:2654-2667

Section II

**Conservation at the
marine foraging grounds**

Chapter 5



Oceanographic habitat of an endangered Mediterranean Procellariiform: implications for marine protected areas

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Ecological Applications (in press)

5.1 Abstract

Marine protected areas (MPAs) require ecologically-meaningful designs, capable of taking into account the particularities of the species under consideration, the dynamic nature of the marine environment, and the multiplicity of anthropogenic impacts. MPAs have been most often designated to protect benthic habitats and their biota. Increasingly, there is a need to account for highly-mobile pelagic taxa, such as marine birds, mammals and turtles, and their oceanic habitats. For breeding seabirds foraging from a central-place, particular attention should be paid to distant foraging grounds

and movement corridors, which can often extend to hundreds of kilometres from breeding colonies. We assessed the habitat use by the most threatened Mediterranean seabird, the Balearic shearwater *Puffinus mauretanicus*, using vessel-based surveys during the chick-rearing period (May - June). We used a hierarchical modeling approach to identify those environmental variables that most accurately reflected the oceanographic habitat of this species by (1) delineating its foraging range using presence / absence data and (2) identifying important foraging grounds where it concentrates in dense aggregations. The foraging range comprised the frontal systems along the eastern Iberian continental shelf waters (depth < 200 m) and areas close to the breeding colonies in the Balearic Islands. Shearwaters aggregated in productive shelf areas with elevated chlorophyll *a* concentrations. Following the model of a core - buffer MPA, we envisioned those areas of dense aggregations (i.e., the area of influence of the Ebro River discharge and Cape La Nao region) as the core regions deserving elevated protection and more stringent management. More diffuse protective measures would be applied within the larger buffer region, delineated by the foraging range of the species. Marine zoning measures can greatly benefit the conservation of the Balearic shearwater and other far-ranging seabirds by extending protective measures beyond their breeding colonies during both the breeding and non-breeding seasons.

5.2 Introduction

Spatially explicit conservation measures are increasingly taking the central stage of fisheries management and marine conservation strategies (Pikitch et al. 2004, Norse et al. 2005). Networks of marine protected areas (hereafter MPAs) are being used to manage fisheries and to protect threatened species and marine habitats around the globe (Houde 2001, Roberts et al. 2001, Halpern 2003). Although most effort has been focused on sessile and sedentary organisms (e.g., coral reefs, mangroves, reef fishes), there is an increasing interest to extend the application of MPAs to encompass highly-mobile species (e.g., marine mammals, birds, and turtles). These far-ranging marine vertebrates are susceptible to multiple anthropogenic impacts (e.g., fisheries bycatch, oil spills) within their foraging grounds and migratory routes (Anderson et al. 2003, Hooker & Gerber 2004, James et al. 2005). At the same time, because these species are often associated with dynamic oceanographic habitat features, an understanding of these wildlife-habitat associations is critical for evaluating the feasibility and design of

pelagic MPAs (Hooker et al. 1999, Hyrenbach et al. 2000, Norse et al. 2005). Thus, marine zoning strategies for highly-mobile marine organisms will require new ecologically meaningful approaches, in tune with their life history and the dynamic nature of the pelagic environment (Hyrenbach et al. 2000, Planes et al. 2000, Gerber et al. 2005).

In the European Union (EU), Council Directive 79/409/EEC (EEC 1979) for the protection of wild birds (Birds Directive) and Council Directive 92/43/EEC (EEC 1992) for the conservation of natural habitats and wild fauna and flora (Habitats Directive) established the foundation for the creation of a network of terrestrial and marine protected areas to conserve natural habitats and wild flora and fauna. The *Natura* 2000 network envisions Special Protection Areas (SPAs, under the Bird Directive) and Special Areas of Conservation (SACs, under the Habitats Directive) to protect those species and habitats in danger of disappearing, while higher protective measures are pending. Within this framework, several studies have quantified the abundance and distribution of cetaceans and sea turtles in the Western Mediterranean (Bearzi et al. 2003, De Segura et al. 2003, Cañadas et al. 2005, Cardona et al. 2005). While little attention has been paid to seabirds in the past, a regional strategy is currently underway for important bird area (IBA) delineation in the Western Mediterranean. This program, under the auspices of BirdLife International, seeks to identify, document, and conserve sites that are key for the long-term viability of bird populations (SPEA-SEO/BirdLife 2005). In the EU, IBAs form the scientific basis and the ornithological reference for the creation of a network of Special Protection Areas (SPAs) under the Birds Directive (Williams et al. 2005). IBAs is a non-government, globally applicable construct and carries no legal obligation, whereas the SPA concept is a legally binding obligation on EU member states.

In this paper, we assess the habitat use by the most threatened Mediterranean seabird, the Balearic shearwater *Puffinus mauretanicus*, as a priority in the design of marine zoning strategies to protect seabirds in this biogeographic region. This shearwater is currently listed as Critically Endangered on the IUCN Red List (Arcos & Oro 2004, Baillie et al. 2004) due to its restricted breeding range (constrained to the Balearic Islands) and small (ca. 2000 breeding pairs; Ruiz & Martí 2004) and declining (7.4% decrease per year; Oro et al. 2004) population size. Population models predict a decrease of 98% within 54 years (three generations), with a mean predicted time of extinction of 40.4 years (Oro et al. 2004). The species is known to feed extensively on

shoals of small pelagic fish, also consuming considerable amounts of fishery discards (Arcos & Oro 2002). Nevertheless, its ecology at sea is still poorly understood (Arcos et al. 2000). In accordance, while much conservation effort is being focused on the breeding colonies, the potential threats at sea (e.g., bioaccumulation of pollutants, fisheries bycatch, and oil spills) have been largely overlooked (Arcos & Oro 2004).

We characterized the oceanographic habitat of the Balearic shearwater in the Western Mediterranean during the breeding season (May – June). Our objective was to assess whether marine zoning strategies could protect the foraging grounds and movement corridors of this endangered species during this critical period of its annual cycle. Following the *Natura 2000* directive, we regarded the habitat as “an environment defined by specific abiotic and biotic factors, in which the species lives at any stage of its biological cycle”. Thus, our habitat-use analysis considered both abiotic (e.g., bathymetric and hydrographic features) and biotic (e.g., ocean productivity patterns) features. We used a hierarchical modeling approach to identify those environmental variables that most accurately reflected the habitat of the Balearic shearwater by (1) delineating the range of the species using presence / absence data and (2) identifying important foraging areas where the species concentrates in dense aggregations. Within this framework, we make recommendations for the implementation of MPAs to protect this critically endangered Western Mediterranean seabird during the chick-rearing season.

5.3 Material and Methods

Study area

We studied the distribution and oceanographic habitat of the Balearic shearwater in the Western Mediterranean during the 1999, 2000, and 2002 chick-rearing period (May - June). Surveys were conducted onboard the R/V “Cornide de Saavedra”, during the annual MEDITS (Mediterranean International Trawl Survey) trawl surveys along the Mediterranean coast of the Iberian Peninsula (Bertrand et al. 2002), from the Strait of Gibraltar in the SW to Cape Creus in the NE (Figure 5.1).

The study area encompassed 2,134 km², with 72 % of the area corresponding to the continental shelf (depth < 200m) and the rest covering the continental slope down to a depth of 1930 m (Figure 5.1). Overall, the continental shelf is very narrow (~ 3 km) in

the Alboran Sea and the Vera Gulf, and widens to the north, reaching a width of up to 70 km in the Ebro Delta - Columbretes area.

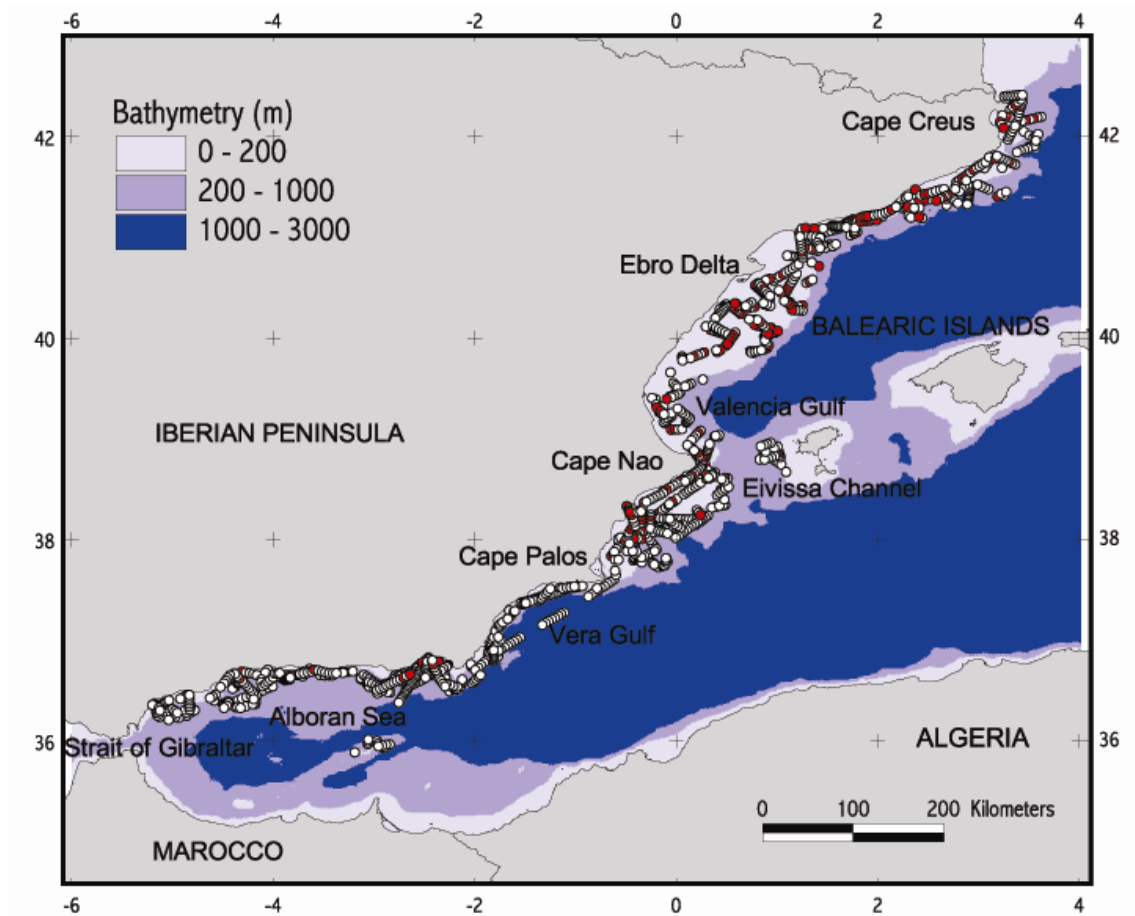


Figure 5.1 Map of the study area showing all sample units (10-minute survey bins; 1999-2002). Geographic location are in °N and °E of latitude and longitude, respectively Red and white dots represent presence and absence of Balearic shearwaters, respectively. Geographic references mentioned in the text are indicated.

Oceanographic context

In the Western Mediterranean, ocean productivity is notoriously heterogeneous, being influenced by marine currents and the input of freshwater run-off, which deliver large amounts of nutrients and fertilize the coastal waters (Arnau et al. 2004). The advection of Atlantic Ocean water, which enters through the Strait of Gibraltar, forms a quasi-permanent-anticyclonic (clock-wise) gyre in the Alboran Sea (Millot 1999). Farther east, the ‘Almería-Oran’ Jet flows from the Iberian coast towards Algeria, giving rise to eddies (Millot 1999). These hydrographic processes promote nutrient enrichment of surface waters in this area (Estrada 1996). Farther north, the Northern Current (or ‘Liguro-Provenço-Catalan’ Current), which originates in the east of the Ligurian Sea

and flows south of the Eivissa Channel along the continental slope, transports rich nutrient waters southwards (Millot 1999, Arnau et al. 2004). In the Ebro Delta area, strong and cold winds from the north and northwest lead to enhanced mixing, the upwelling of nutrients, and the consequent increase of plankton production (Estrada 1996, Salat 1996). A permanent front with strong salinity gradients, known to limit the seaward dispersion of the larvae of coastal and shelf species (i.e., fish, crustaceans), retains this high localized productivity near the coast (Font et al. 1988, Sabatés 1996, Abelló & Guerao 1999). This productive region represents a major fish spawning area, especially for small pelagic species like anchovy and sardine (Sabatés 1996, Salat 1996, Agostini & Bakun 2002), and supports one of the most important fishing fleets of the Western Mediterranean (Demestre et al. 1988, Estrada 1996, Salat 1996). Additionally, submarine canyons north of Barcelona are also thought to enhance ocean productivity (Palanques et al. 2005).

At-sea surveys

All observations were made by the same observer (JMA) during 1999, 2000, and 2002 using standardized strip-transect techniques (Tasker et al. 1984) adapted to match the specific conditions to the study area. A 300 m strip-width transect band was used, with the observer surveying both sides of the vessel (i.e., 600 m band) only when conditions (visibility and wind) were appropriate. The width of the band was checked periodically using a range-finder (Heinemann 1981). Snap-shot counts were used to census flying birds (Tasker et al. 1984). All Balearic shearwaters observed within the survey transect were recorded. Additionally, the number of trawlers sighted within a three-nautical-mile (5.58 km) band were also recorded, to account for the influence of fishing discards on the distribution of the species. The shearwater and vessel sightings were summed up into 10-minute survey bins.

Environmental variables

Sea surface temperature (SST hereafter) and chlorophyll *a* concentration (chl *a* hereafter) are useful proxies of water mass distributions, frontal systems, and ocean productivity domains (e.g., LeFevre 1986, Longhurst 1998), and have been previously used to characterize the oceanographic habitats of seabirds (e.g., Spear et al. 2001, Hyrenbach et al. 2002). We relied on remotely-sensed environmental information,

which is available at a variety of spatial and temporal resolutions, to quantify SST and chl *a* conditions during our seabird surveys. We used the 8-day composites of 9-km pixel data because the imagery at the finer daily temporal resolution was obstructed by cloud cover to a great extent.

We used night-time Pathfinder 4.1 SST imagery from the Advanced Very High Resolution Radiometer (AVHRR), to quantify water temperature during our surveys (http://podaac.jpl.nasa.gov/pub/sea_surface_temperature/avhrr/pathfinder/data_v4.1/). We derived the chl *a* concentrations from Sea-viewing Wide Field-of-view Sensor (SeaWiFS) imagery (<http://seawifs.gsfc.nasa.gov/SEAWIFS.html>). We discarded any values beyond the range of SeaWiFS validation (0.05 - 50 mg m⁻³; Hooker & McClain 2000). We considered four different productivity regimes based on the chl *a* values: oligotrophic (low nutrient, chl *a* < 0.1 mg m⁻³), mesotrophic (intermediate nutrient, chl *a*: 0.1 - 0.3 mg m⁻³), eutrophic (high nutrient, chl *a*: 0.3 - 1 mg m⁻³), and enriched waters (chl *a* > 1 mg m⁻³) (Kahru & Mitchell 2000).

As water depth influences seabird distributions, as indicated by the different communities that inhabit distinct coastal and pelagic domains (Schneider 1997), we included this variable in the habitat model. We obtained bathymetric data from NOAA's ETOPO 5-minute (spatial resolution of 7.5 - 8.2 km at 35° N and 42° N latitude, respectively) gridded elevation dataset (NGDC 1998).

Because these three habitat variables (SST, chl *a* and depth) were not normally distributed, we used the median as a conservative statistic to express the central tendency of these variables within each 9-km cell grid (Zar 1999). In addition, we quantified the spatial gradients of these properties for each 9-km grid cell. We calculated the proportional change (PC) in SST, chl *a*, and depth within a surrounding 3 x 3 pixel (27 x 27 km) grid as follows: $PC = [(maximum\ value - minimum\ value) * 100] / (maximum\ value)$. This dimensionless metric expresses the magnitude of change in each habitat variable, scaled to the maximum value.

To account for the influence of colonies on the distribution of central-place foraging seabirds at-sea during the breeding season, we included the distance from the centroid of each grid cell to the nearest breeding site (see Louzao et al. 2006 for colony locations). Finally, we also included the distance from each grid cell to the nearest shoreline in the model to account for additional potential onshore - offshore patterns.

Data processing

We explored shearwater habitat associations using the ArcView 3.2 Geographic Information Systems (GIS) software (1999), by merging the seabird distribution data with the environmental information. We first standardized the survey effort to account for differences in the area surveyed during each 10-min bins, due to varying cruising speeds and strip widths. We aggregated the survey bins into 9-km grid cells (81 km²), at the same spatial scale of the remote sensing imagery, and recalculated the occurrence and density of shearwaters and trawlers for these grid cells. We repeated this process for each 8-day survey period, to match the temporal scale of the remote sensing imagery. Thus, whenever a grid cell was visited repeatedly in the course of the same cruise, it contributed a data point for each 8-day period it was surveyed.

We then extracted the oceanographic variables (SST, chl *a*, depth) for each grid cell using the WIM 6.26 software (Kahru 2000). Because the median and gradient values for each grid cell were calculated within a surrounding 3 × 3 pixel (27 × 27 km) neighborhood, the number of valid data points could range from 0 to 9; we rejected those grids cells with fewer than 2 valid data points. After standardization, we calculated the distance between the centroid of each grid cell and the nearest shearwater colony and shoreline using the ArcView 3.2 GIS.

Data analysis

Spatial autocorrelation. Before undertaking the statistical analysis of the Balearic shearwater distributions, we had to ensure that the observations were independent (i.e., the number of birds in any grid cell was not related to their distribution in adjacent grid cells). In clumped distributions, like seabird flocks, adjacent observations are often more similar than would be expected in randomly distributed data, revealing significant positive autocorrelation patterns (Schneider 1990, Sokal et al. 1998). Significant autocorrelation can cause the false rejection of the null hypothesis (Type I error), giving raise to spurious significant habitat relationships (Hurlbert 1984). We used the Moran's *I* coefficient to assess the spatial autocorrelation patterns in the seabird data. This index, developed to quantify spatial patterns for highly non-normal distributions, ranges from -1 (negative autocorrelation) to +1 (positive autocorrelation), with null values being indicative of a lack of spatial structure (Sokal & Oden 1978).

We quantified the similarity of the Balearic shearwaters occurrence (presence / absence) and density (birds 100 km⁻²) for all pair-wise combinations of grid cells as a function of their spatial distance, and considered up to 15 spatial lags at 9-km intervals (0 - 135 km) in any direction (i.e., assuming an isotropic spatial autocorrelation structure). We assessed the significance of the Moran's *I* values using Monte Carlo randomization tests (Manly 1994). Both the Moran's *I* values and randomization tests were performed using the Rook Case V.1.6 software (Sawada 1999). Briefly, we performed 1500 permutations at each lag distance and contrasted the observed Moran's *I* value with those calculated by the randomization procedures. The Bonferroni correction was used to account for multiple testing.

Model building. We related shearwater occurrence (presence / absence) and density (when shearwaters were present) to eleven explanatory variables. We used eight environmental variables: median depth, depth gradient, distance to shoreline, distance to the closest colony, median SST, SST gradient, median chl *a*, and chl *a* gradient. Due to unequal survey effort in each grid cell, we included survey effort (km²) as a covariate in the analysis to account for potential biases. Also, we included the occurrence and density of trawlers because of the potential influence of fishery discards on shearwaters distribution. Finally, we considered the effect of "year" to assess interannual variability in shearwater distribution and abundance during our study.

Because the environmental conditions in the study area were cross-correlated (Table 5.1), we used multivariate statistics to assess the relative influence of each habitat variable on shearwaters distribution. We used generalized linear models (GENMOD procedure; SAS Institute 2000), and adopted a hierarchical modeling approach, where we first characterized the overall range of the species using occurrence (presence / absence) records from the entire dataset. We built this model assuming a binomial distribution and using the logit-link function, and evaluated the goodness-of-fit using the Pearson's chi-square statistic (Crawley 1993). Then, after excluding those bins where shearwaters were absent, we quantified the areas of high aggregation using the continuously distributed density data using the identity link function.

Table 5.1 Bivariate Spearman rank P values and correlations between environmental variables in the analyzed 9-km grid cells ($n = 567$). NS = non-significant; P* (<0.05); P** (<0.01); P*** (<0.001).

Variable	Distance shoreline	Distance colony	Median depth	Depth gradient	Median chl <i>a</i>	Chl <i>a</i> gradient	Median SST	SST gradient
Distance shoreline	1.000	P***	P***	P***	P***	NS	NS	P*
Distance colony	-0.271	1.000	P***	P**	P***	NS	P***	P**
Median depth	0.323	0.415	1.000	P***	P*	P*	P***	P***
Depth gradient	-0.668	0.124	-0.381	1.000	P***	NS	NS	P***
Median Chl <i>a</i>	-0.196	0.419	-0.083	0.153	1.000	P***	P***	P***
Chl <i>a</i> gradient	0.024	-0.025	-0.095	0.001	0.370	1.000	P**	NS
Median SST	0.057	-0.384	-0.178	-0.017	-0.462	-0.122	1.000	P***
SST gradient	0.107	0.110	0.167	-0.169	0.155	0.057	-0.180	1.000

We assembled the best-fit model iteratively using a stepwise procedure by rejecting/retaining the variables with the lowest/highest explanatory power one at a time (variables were discarded at $\alpha > 0.05$, and retained otherwise). Then, we calculated the probability of shearwater occurrence for each surveyed grid cell in each year, using the final logistic model developed with the presence / absence data. We calculated the Linear Predictor (LP) using the constant and the variable coefficients from the best-fit model, and calculated the probability of shearwater occurrence predicted by the model (P) as follows: $P = e^{LP} / (1 + e^{LP})$ (Crawley 1993).

Suitable habitat. The receiver operator curve (ROC) plots provide a powerful approach to assess the performance of predictive ecological models based on presence / absence data (van Erkel & Pattynama 1998, Pearce & Ferrier 2000). This graphical method portrays the trade-off between the “sensitivity” (percentage of presence events correctly predicted) and “1 - specificity” (percentage of the absence events correctly predicted) of the predictive model (van Erkel & Pattynama 1998; Boyce et al. 2002). Moreover, the area under the ROC curve (AUC) provides a metric of the predictive power of the model (Thuiller et al. 2003), which ranges between 0.5 (null predictive power, corresponding to the line of 45 °) and 1 (a perfect model) (Boyce et al. 2002). We calculated the sensitivity and specificity of the model for a range of probabilities, and calculated the corresponding AUC value using the approach of Watkins (2000). We

identified the probability value, which maximized the sensitivity and specificity of the model predictions using the ROC and AUC results (Pearce & Ferrier 2000, Boyce et al. 2002, Pearson et al. 2004). This threshold value was determined by the probability level where the sensitivity and the specificity curves intersected (Thuiller et al. 2003, Pearson et al. 2004).

Table 5.2 Summary of seabird surveys, showing the dates and survey effort during MEDITS cruises in 1999, 2000, and 2002. Effort is expressed as both the area (km²) and the number of 9-km grid cells analyzed.

Year	Cruise dates	Surveyed area (km ²)	Number of grid cells
1999	4 May - 4 June	588.6	211
2000	22 May - 22 June	571.6	163
2002	11 May - 13 June	612.5	193
Total	95 days	1772.6	567

5.4 Results

At-sea surveys

We surveyed a total of 1409 10-min bins, which we aggregated into 680 grid cells during 95 survey days spanning the three years of our study (Table 5.2). Due to cloud cover, we discarded 16.6 % of the surveyed grid cells with incomplete remote sensing information. We used the remaining grid cells ($n = 567$) for the analysis of shearwater occurrence, and those grids with shearwaters presence ($n = 140$) for the analysis of density. We sighted 657 shearwaters over three years, with their maximum densities varying between 900 and 3453 birds 100 km⁻² from year to year (Table 5.3). Overall, most of the shearwater sightings occurred between Cape Creus (42.3° N) and the vicinity of Cape Palos (38.0° N) (Figure 5.1).

Table 5.3 Summary of the Balearic shearwater observations, showing the total number of birds sighted, the number and proportion of occurrence, and the maximum density during each survey year. The occurrence and density statistics are derived from the 9-km grid cells used for analysis.

Year	Number Recorded	Occurrence (Number / Proportion)	Maximum Density (birds 100 km ⁻²)
1999	387	56 / 26.5	3453
2000	175	37 / 22.7	1675
2002	95	47 / 24.3	900
Total	657	140 / 24.7	

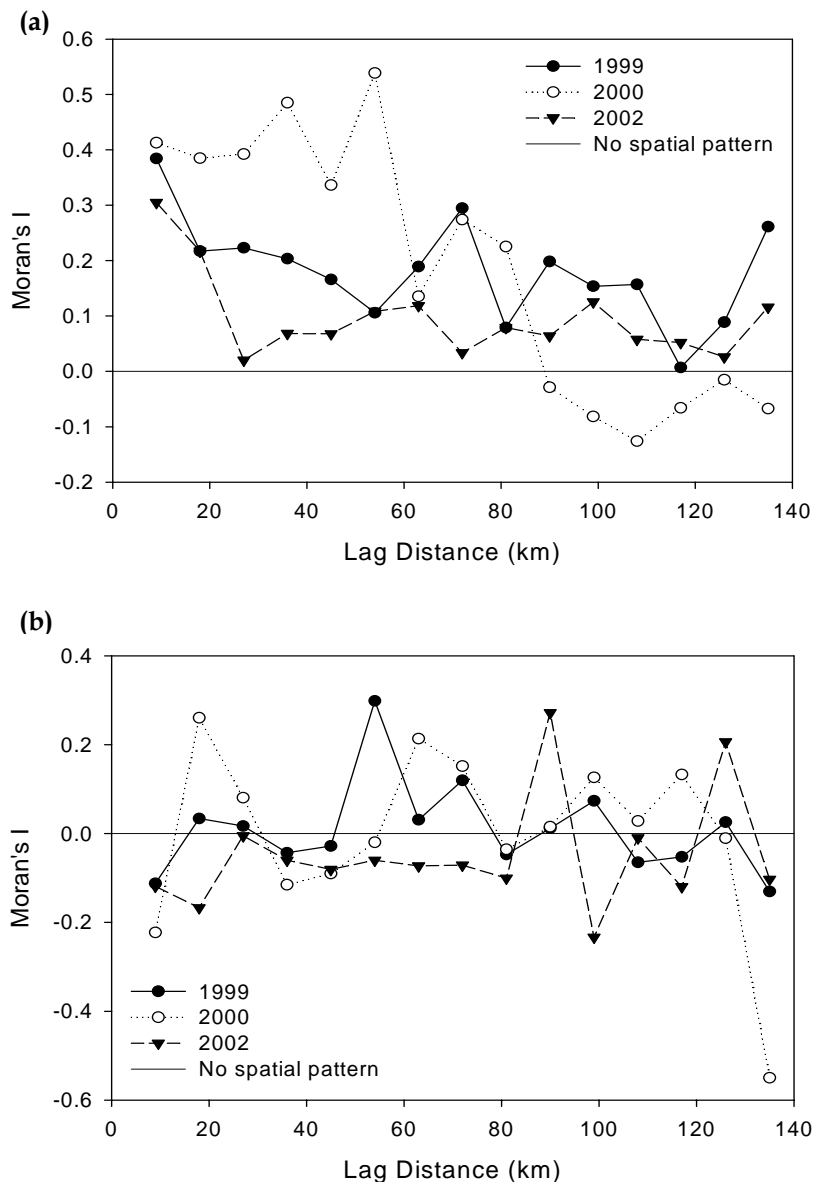


Figure 5.2 Analysis of spatial autocorrelation: Moran's I correlogram for Balearic shearwater (a) occurrence and (b) density during the three years of surveys (1999-2002). The solid, horizontal line at zero represents a baseline showing no spatial pattern. All Moran's I values were non-significant.

Spatial autocorrelation

We found no evidence of significant spatial autocorrelation in the at-sea distribution of the Balearic shearwater, neither for the occurrence nor the density data. These results indicated that the 9-km grid cells yielded independent observations, and thus provided a suitable spatial scale of analysis (Figure 5.2). Both the occurrence and density correlograms did not show significant patterns. However, the occurrence correlogram suggested some degree of aggregation up to fairly large spatial scales (54-63 km) in 1999 and 2000, and at a smaller spatial scale (18-27 km) in 2002. Overall, the shearwater distribution pattern seemed more spatially constrained in 2000, with larger positive Moran's I values at smaller lags (9-54 km) and negative values at larger lags (> 90 km). Conversely, all Moran's I values were positive in 1999 and 2002 (Figure 5.2a). For the density data, all the Moran's I values were close to 0, suggesting a random distribution (Figure 5.2b).

Association to oceanographic habitats

Balearic shearwater occurrence. The analysis of the presence / absence data identified seven significant predictor variables: distance to the nearest colony and shoreline, median depth, depth gradient, chl a gradient, survey effort, and year of survey (Table 5.4). The interpretation of these results is based on the positive or negative sign of the estimated response coefficients, where a positive or negative sign is indicative of a greater or smaller probability of encountering a shearwater in a given grid cell, respectively. Therefore, the negative signs of the distance to the nearest colony and shoreline, the median depth, and depth gradient were suggestive of the distribution of the species along the Iberian continental shelf, close to the coast and to breeding colonies. The positive sign of the chl a gradient estimate suggested the association of the Balearic shearwater with oceanographic fronts indicative of productive coastal waters and river plumes. The positive association with the survey effort suggested that the greater the area surveyed within a 9-km grid cell, the greater the probability of encountering a shearwater. Because the interannual effect was calculated relative to the last year of the study, the encounter probability was higher in 1999 and 2000 than in 2002 (see the null value of 2002 listed in Table 5.4). Altogether, the best-fit model

explained 21 % of the observed deviance. The goodness-of-fit test indicated that the model performed well ($\chi^2_{558} = 579.00, P = 0.74$).

Once we identified the significant variables, we represented the shearwater distribution and the underlying environmental datasets graphically using GIS. Balearic shearwaters occurred predominantly around the Ebro Delta and in the vicinity of Cape La Nao, especially in 1999 and 2000. In 2002, the shearwaters showed a more widely scattered pattern, with sightings from Cape Creus to Cape Palos (Figure 5.3). We superimposed the shearwater distributions over the average chl *a* gradients for each year using the monthly data from those months which better matched the temporal windows of the annual surveys: May (1999 and 2002) and June (2000) (cf. Table 5.2). We used the chl *a* monthly average data only for mapping because they provided a more complete (i.e., cloud free) spatial coverage of the study area. However, the habitat-use models used the finer-scale satellite imagery exclusively (see Material and Methods).

Table 5.4 Results of the generalized linear model (GLM) analysis of the Balearic shearwater occurrence during the three years of this study. Only significant variables are shown. Degrees of freedom (DF), chi-square, and significance are given for the global contribution of the year effect and not for each specific year since the year variable is categorical and calculated relative to the last year of study (see the null value of 2002 listed). S.E.: standard error.

Variable	Estimate parameter	S.E.	D.F.	Chi-square	Significance
Distance to colony	- 0.007	0.001	1	33.82	< 0.0001
Distance to shoreline	- 0.026	0.012	1	4.69	0.030
Median depth	- 0.003	0.001	1	75.22	< 0.0001
Depth gradient	- 0.020	0.006	1	4.38	0.0364
Chl <i>a</i> gradient	0.015	0.005	1	5.31	0.021
Survey effort	0.232	0.062	1	5.54	< 0.019
Year (1999)	0.639	0.269	2	7.39	0.025
Year (2000)	0.037	0.288			
Year (2002)	0.000	0.000			

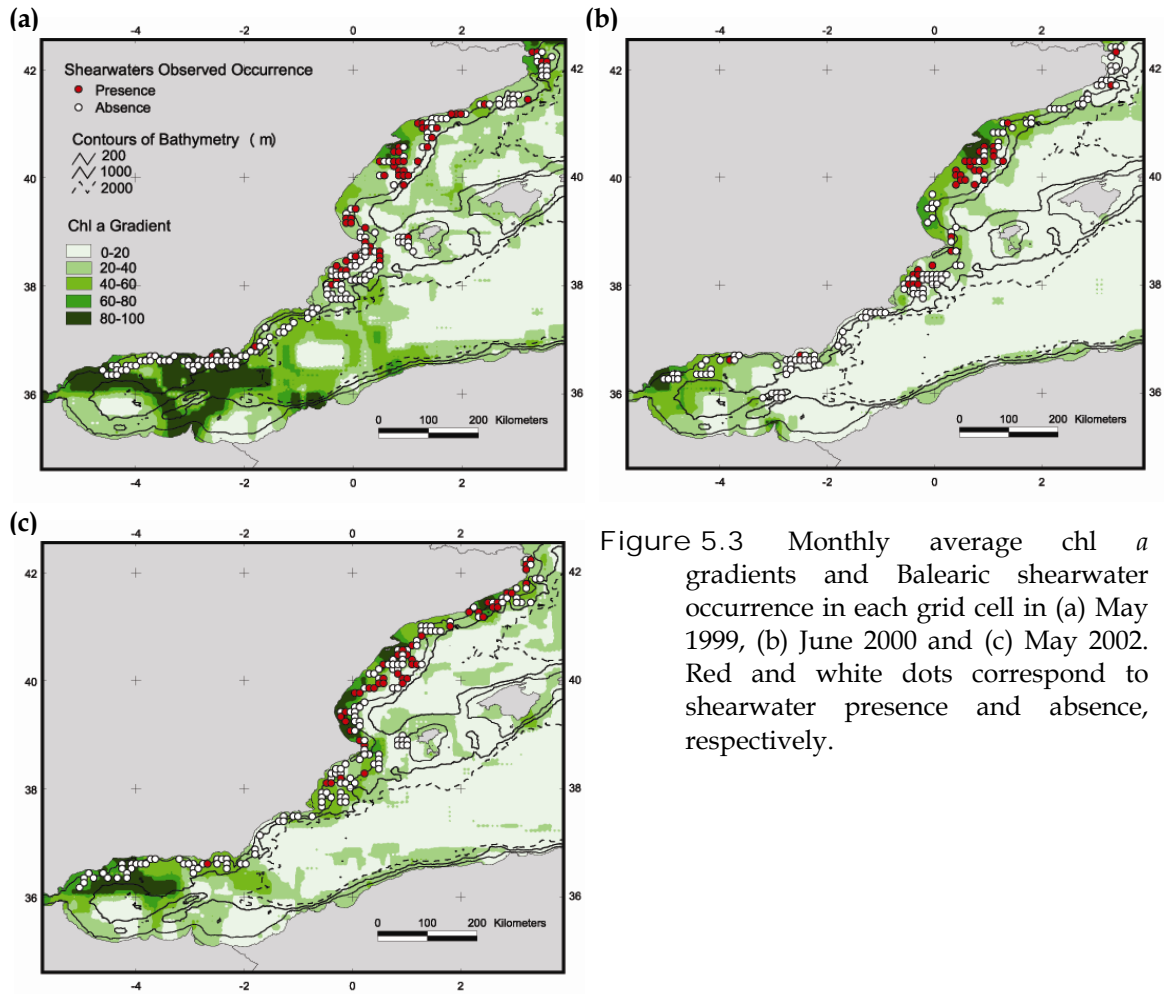


Figure 5.3 Monthly average chl *a* gradients and Balearic shearwater occurrence in each grid cell in (a) May 1999, (b) June 2000 and (c) May 2002. Red and white dots correspond to shearwater presence and absence, respectively.

Testing model accuracy and predictive power. We obtained an AUC value (\pm SE) of 0.795 (\pm 0.006) (Figure 5.4a), which was indicative of a good model performance (Swets 1988). The model accurately classified 71.4 % of the presences and 72.6 % of the absences of Balearic shearwaters in 9-km grid cells. When we intersected the sensitivity and specificity curves, we identified a probability threshold of 0.3 (Figure 5.4b). Thus, we considered those 9-km grid cells where the predicted encounter probability exceeded 0.3 as the suitable habitat for the Balearic shearwater.

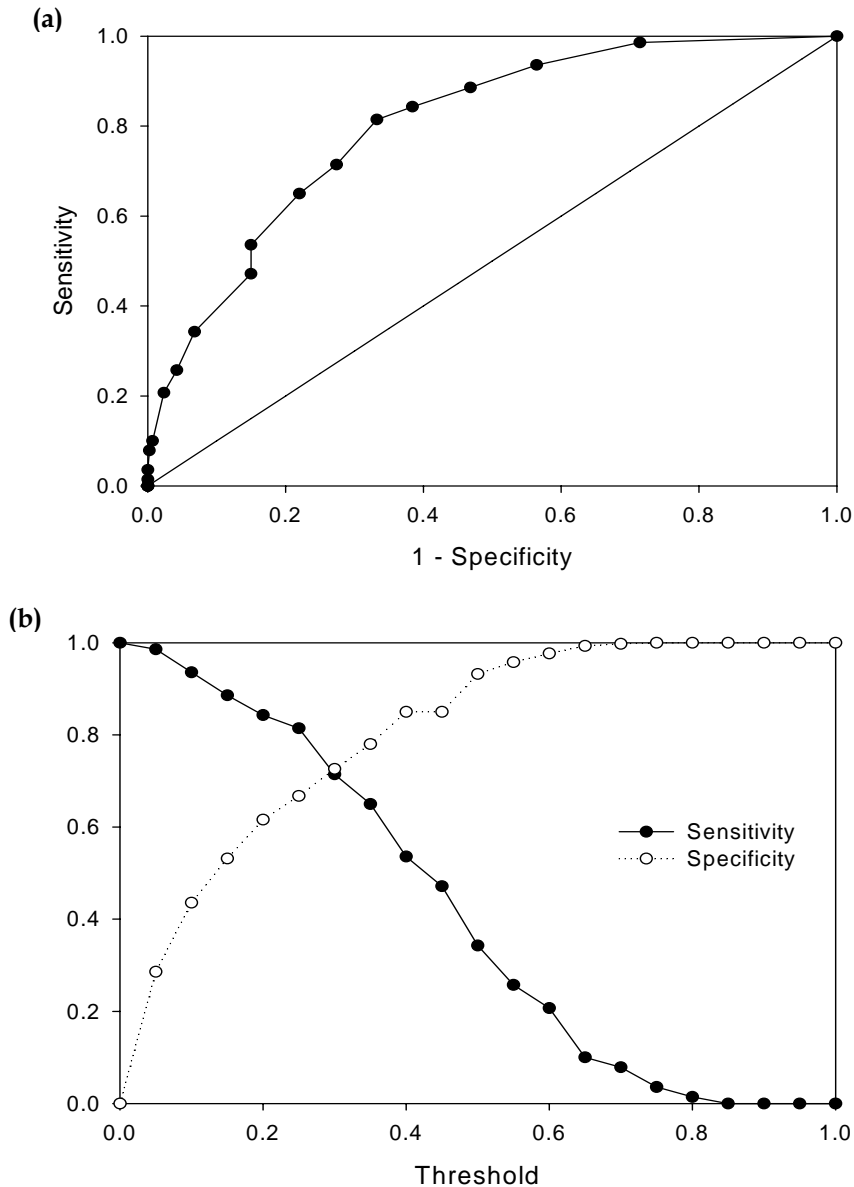


Figure 5.4 Receiver operator curve (ROC) curves showing the trade-offs when (a) the sensitivity is plotted vs. 1 - specificity, where the 1:1 line represents the expected probabilities if shearwaters were randomly distributed, and (b) the sensitivity and the specificity are calculated for each probability level to identify the suitable oceanographic habitat threshold ($Pr = 0.3$ in this case) (see details in Material and Methods: Data analysis: Suitable habitat).

We generated maps of suitable habitat for each year, extending the area of interest to the northwestern African coast and including the breeding colonies in the Balearic archipelago (Figure 5.5). We used the seven significant variables identified by the model to recalculate occurrence probabilities within this expanded area for each year. From the centroid of each grid cell we computed the four variables that did not vary between years: distance to the nearest colony and shoreline, median depth, and depth gradient. To account for biases in survey effort, we calculated an average survey effort

for the entire study period and applied that same value to all the grid cells. However, the monthly average values of the chl *a* gradient and the year coefficients varied over time, driving the interannual variability in the location and extent of the suitable oceanographic habitat.

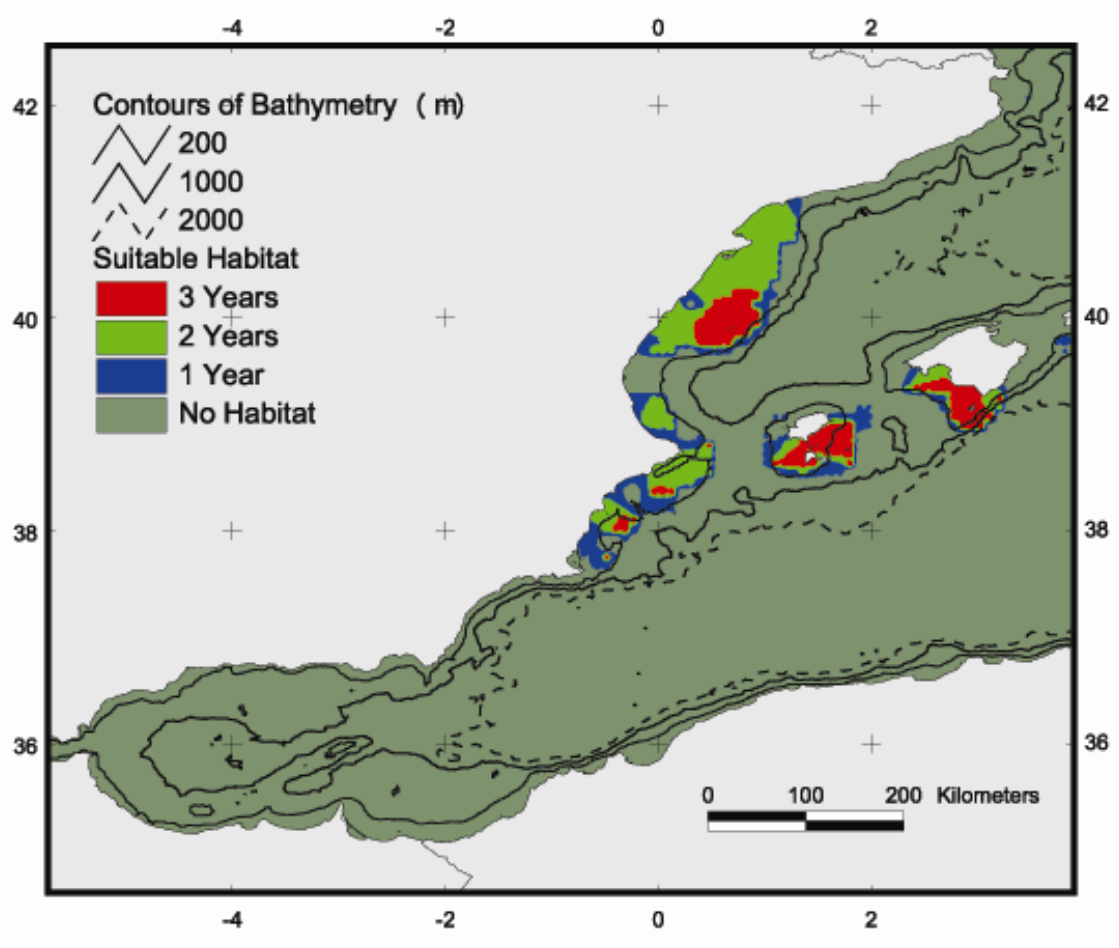


Figure 5.5 Predicted suitable oceanographic habitat of the Balearic shearwater modeled using three years of data (1999, 2000, 2002) and the threshold value of $Pr = 0.3$ (see Material and Methods: Data analysis: Model building). Each 9-km grid cell is colour coded to show that it was identified as suitable habitat in one year (blue), two years (green) or three years (red).

For the entire study period (1999-2002), the suitable oceanographic habitat included, in general terms, two areas in the Iberian continental shelf: the Ebro Delta area in the north and Cape La Nao in the south (latitudinal limit of 38.0° N); and waters close to the breeding colonies at the Balearic archipelago. These areas were classified as suitable shearwater habitat in at least two of the three survey years (Figure 5.5).

Balearic shearwater density. The analysis of shearwater densities identified three significant predictor variables: median chl *a*, survey effort, and year (Table 5.5). The positive effect of median chl *a* suggested that shearwaters were more abundant in

highly productive waters. The survey effort revealed a sampling bias, with apparent shearwater densities within a grid cell diminishing with increasing survey area. Thus, shearwater sightings in grids with low survey effort tended to yield very high apparent density values. Finally, the year effect suggested that shearwater densities were higher during the first two years of the study (1999, 2000) than in 2002. Altogether, the best-fit model explained 15 % of the observed deviance.

Table 5.5 Results of generalized linear model (GLM) analysis of the Balearic shearwater density during the three years of this study. Only significant variables are shown. Interannual effect was calculated relative to the last year of study (see the null value of 2002 listed). S.E.: standard error, D.F.: degrees of freedom.

Variable	Estimate parameter	SE	D.F.	Chi-square	Significance
Chl median	287.490	118.490	1	6.88	0.009
Survey effort	- 48.204	15.978	1	8.90	0.003
Year (1999)	193.622	73.868	2	6.79	0.034
Year (2000)	82.693	81.557			
Year (2002)	0.000	0.000			

To illustrate the spatial distribution of shearwater aggregations, we plotted their density over the extent of different productivity regimes (Kahru & Mitchell 2000), defined using the monthly averages of median chl *a* for each year separately (Figure 5.6). The study area was characterized by a general mesotrophic regime (intermediate productivity). Some continental shelf locations were annually characterized by eutrophic (high nutrient) and some highly enriched areas. The area north of Cape Creus was highly productive in 1999, and the area around the Ebro Delta was characterized by elevated chl *a* concentrations in 2000 and 2002 (Figure 5.6).

The observed maximum values of shearwater density diminished over the study period (Table 5.3) and the distribution patterns displayed noticeable interannual variability (Figure 5.6). In 1999, two main areas of shearwater aggregations were identified, in both cases associated with mesotrophic conditions. The area around the Ebro Delta supported the highest densities, while the area around Cape La Nao was characterized by lower densities (Figure 5.6a). In 2000, shearwaters were more strongly concentrated off the Ebro Delta, associated with the highest (highly eutrophic) chl *a* levels (Figure 5.6b). However, in 2002, shearwaters were widely distributed across the Iberian continental shelf from Cape Creus to the latitudinal limit of 38.0 ° N. We did

not identify any areas of dense foraging aggregations, even though the Ebro Delta region registered the highest chl *a* levels in 2002.

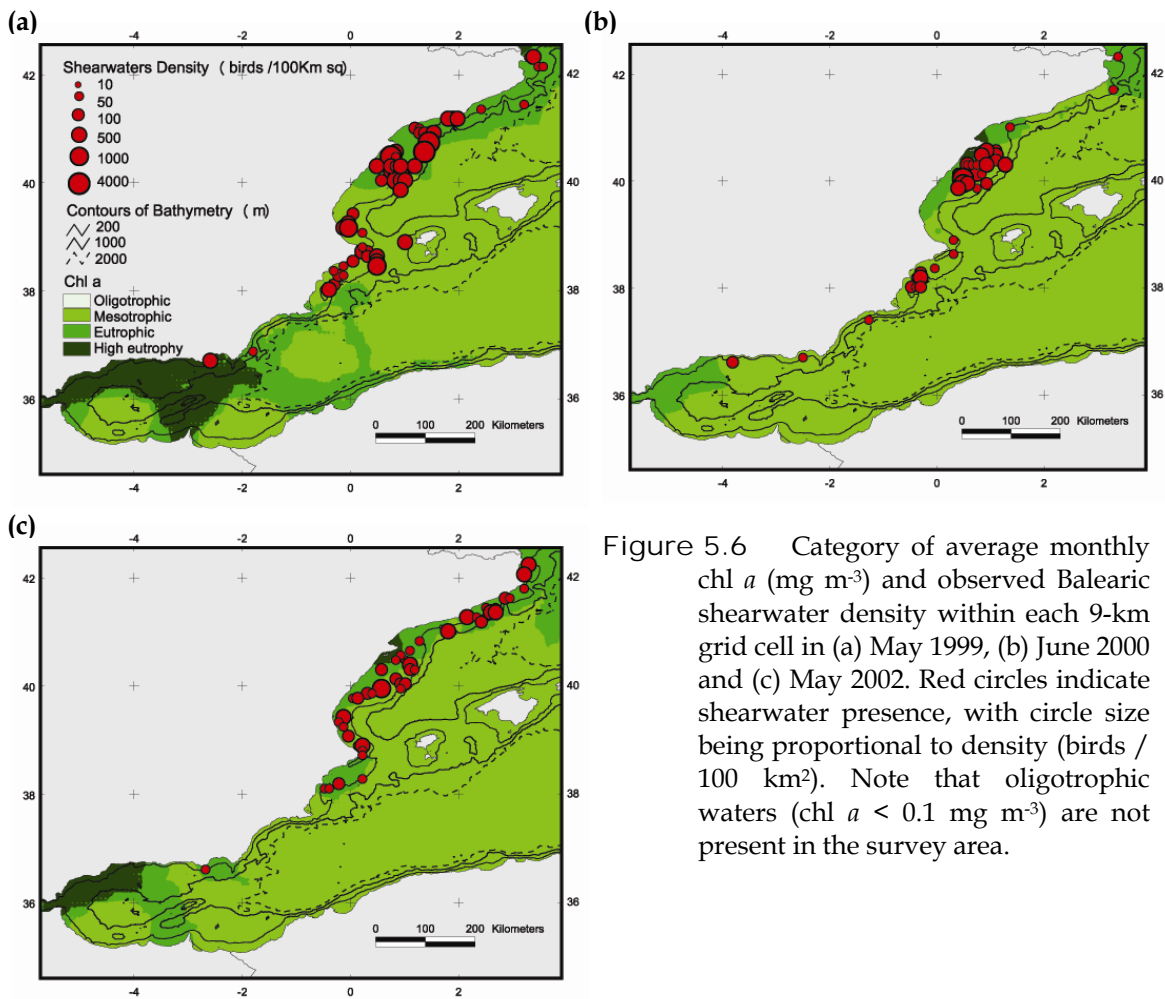


Figure 5.6 Category of average monthly chl *a* (mg m^{-3}) and observed Balearic shearwater density within each 9-km grid cell in (a) May 1999, (b) June 2000 and (c) May 2002. Red circles indicate shearwater presence, with circle size being proportional to density (birds / 100 km^2). Note that oligotrophic waters ($\text{chl } a < 0.1 \text{ mg m}^{-3}$) are not present in the survey area.

5.5 Discussion

Oceanographic habitat of the Balearic shearwater

Although the Mediterranean is considered an oligotrophic sea, local ocean productivity is considerably variable in time and space (Estrada 1996, Agostini & Bakun 2002, Arnau et al. 2004). For instance, the remotely-sensed chl *a* values documented in the study area were generally characteristic of a mesotrophic regime (Kahru & Mitchell 2000), even though the highly productive areas of the Ebro Delta and the Alboran Sea gyres were characterized by highly enriched phytoplankton pigment concentrations. The foraging range of the Balearic shearwater during the chick-rearing season mainly comprised the Iberian continental shelf between 42.3° N

and 38.0 ° N. These analyses revealed that shearwaters were present in shallow shelf and coastal waters characterized by frontal systems in areas closer to their breeding colonies. Within this larger foraging range, shearwaters were most numerous in the highly productive waters with elevated chl *a* concentrations around the Ebro Delta (especially in 2000) and the area extending to the south of Cape La Nao. These results suggested that the shearwaters were responding to diverse oceanographic processes. They were likely aggregating off the Ebro Delta in response to plumes of higher productivity associated with the spring – summer input of surface nutrients from the Ebro River, during a period of weak vertical mixing (Salat et al. 2000). The shearwater sightings around Cape Creus could also be explained by the presence of mesoscale oceanographic features (Arnau et al. 2004) and submarine canyons (Palanques et al. 2005) acting to increase localized productivity and aggregate prey. Both the distribution and abundance patterns showed substantial interannual variability, which was likely associated with the influence of some additional explanatory variables not included in the models. Namely, year-to-year changes in the discharge of the Ebro River, shifts in the stocks of small pelagic fish, and spatio-temporal variability of the fishing effort and fishery discards, as well as survey effort, could influence shearwaters distribution patterns (Lloret et al. 2001, Agostini & Bakun 2002, Arnau et al. 2004).

The oceanographic habitat model results matched the foraging range described by the observed data, and also identified the waters close to the breeding colonies in the Balearic Islands as potentially suitable habitat for the Balearic shearwater. Conversely, the Alboran Sea was not identified as suitable shearwater habitat, although it was characterized by high primary productivity associated with high chl *a* gradients and hydrographic fronts (Millot & Taupier-Letage 2004). We hypothesize that the large distance between the breeding colonies and the Alboran Sea make these productive waters unsuitable for central-place foraging shearwaters.

Our results agree with previous studies, which have underscored the near-shore distribution of the Balearic shearwater and its tendency to occur in the vicinity of the Ebro Delta (Abelló & Oro 1998, Arcos & Oro 2002, Abelló et al. 2003). Yet, to our knowledge, this is the first time that a significant association of Balearic shearwaters and frontal systems has been documented on the basis of quantitative analyses of oceanographic variables. The association of this endangered species with dynamic oceanographic habitats, defined by the extent of hydrographic features (e.g., frontal

systems, productive river plumes), has important implications for its management and conservation.

In particular, frontal regions where adjacent water masses of different properties come together are often characterized by convergence zones which aggregate marine organisms, including seabirds and their zooplankton and fish prey (LeFevre 1986, Franks et al. 1992, Hunt et al. 1996). Aggregations of upper-trophic-level organisms are presumably the consequence of the tight coupling of predator-prey interactions at frontal systems, due to either enhanced aggregation or availability of sub-surface prey to diving predators (Schneider 1993). Several previous studies have revealed dense aggregations (100 - 1000 birds km⁻²) of diving shearwaters at narrow (~ 10 km) tidal fronts characterized by elevated and persistent productivity within broad continental shelves (Hunt et al. 1996, Begg & Reid 1997). While Balearic shearwaters did not aggregate into such dense flocks, we documented intense aggregations (10 - 40 birds km⁻²) in the vicinity of the Ebro Delta (Table 5.3, Figure 5.6).

As part of our analysis, we did not detect any significant association of shearwaters with trawlers, suggesting that while fishery discards make an important contribution to the energetic requirements of the species (Arcos & Oro 2002), they do not influence their distribution over large spatial scales (10s - 100s km) (cf. Skov & Durinck 2001). Thus, we hypothesize that discards represent a quantitatively important, yet secondary foraging resource, taken opportunistically in the absence of alternative prey (cf. Arcos et al. 2006). In this case, oceanographic conditions were the major determinant of the Balearic shearwater distribution, as has been previously suggested for other opportunistic scavenging seabirds in the North Sea (Camphuysen & Garthe 1997). Nevertheless, since the highly productive areas used by the shearwaters also sustain demersal trawling fleets (Demestre et al. 1988), it is difficult to tease apart the influence of natural (e.g., spawning aggregations) and anthropogenic (e.g., fishery discards) food sources on their distributions.

One of the main contributions of this study to the ecology of the Balearic shearwater is the ability to map potentially suitable habitat in other areas of the Western Mediterranean not studied using ship-based surveys. However, this approach must be taken with caution because it assumes that the habitat associations we defined for the study area can be extrapolated to other unsurveyed areas, such as the Gulf of Lions and the North African Coast. This example illustrates the need for repeated surveys during contrasting oceanographic conditions to validate habitat suitability models

developed for a specific area (Forney 2001, Raymond & Woehler 2003), and shows the value of predictive models as an exploratory tool to identify additional potential habitats of highly endangered species in other poorly studied areas. Suitable habitat areas identified by model results could be targeted for exploratory ship-based and aerial surveys. In both cases, the ability to repeatedly survey protected species distributions and concurrent oceanographic conditions over different seasons and years is critical for developing, validating, and refining habitat suitability models. In particular, these standardized surveys are critical to identify time trends both between seasons (early – late spring) and across years (population trends).

Future research should also focus on obtaining a mechanistic understanding of the small-scale interactions between local oceanographic conditions, prey distribution, and the aggregation patterns and behavior of Balearic shearwaters within the high-use foraging grounds identified in the present study. Promising avenues entail satellite tracking of individual foragers and small-scale surveys along the frontal systems where the species concentrates (e.g., Begg & Reid 1997, Hyrenbach et al. 2002).

Conservation measures: implementation of marine protected areas

The degree of aggregation and the habitat associations of far-ranging seabirds greatly influence their susceptibility to anthropogenic threats, such as longline bycatch and oil spills, and the potential use of marine protected areas to mitigate those threats (Anderson et al. 2003, Hyrenbach et al. 2006). Seabirds are particularly susceptible to anthropogenic impacts at certain time periods (e.g., breeding season) and localities (e.g., foraging grounds) when / where they aggregate in dense concentrations (Hunt et al. 1996, Huettmann & Diamond 2000). Similarly, the feasibility and effectiveness of specific management practices depend on both the spatial extent and the degree of aggregation of the protected species and the threats in question.

Nevertheless, given its delicate status, marine zoning measures may greatly benefit the conservation of the Balearic shearwater by extending protective measures beyond the breeding colonies. In particular, MPAs and time-area closures may help mitigate fisheries bycatch, impacts from wind farms, and the risk of catastrophic oil spills. Habitat models could guide the design of effective, spatially explicit conservation measures by characterizing those areas of special interest for conservation such as foraging areas and movement corridors (e.g., BirdLife International 2004). In principle,

marine zoning strategies based on geographically fixed features (e.g., seamounts, shelf-breaks, submarine canyons) are much more conducive to implementation, because their boundaries are defined by the extent of static depth contours. Dynamic MPA boundaries would be much more difficult, but not impossible, to implement (Hyrenbach et al. 2000, Norse et al. 2005). However, there is a clear need to develop such a MPA model, and marine zoning measures defined by seasonal and interannual changes in water mass distributions have already been used to mitigate fisheries bycatch (e.g., Nagao et al. 1993, NOAA 2002).

Following the model of a core - buffer MPA, we envision the following marine zoning approach to protect Balearic shearwaters during the breeding season. Areas of dense aggregations could serve as the core regions deserving elevated protection. Namely, the area of influence by the Ebro River discharge and Cape La Nao region would be ideal candidates for stringent protective measures, including prohibition of oil tanker traffic and the adoption of adequate measures to avoid shearwater fisheries bycatch (Arcos & Oro 2004, Arcos et al. 2006). The dynamic nature and interannual variability of fronts raises the necessity of the periodical revision of their geographic extension. Nevertheless, these features could be monitored remotely using satellite imagery and certain areas of likely habitat could be delineated on the basis of the probability of occurrence of these frontal systems. Other areas identified by the modeling of suitable habitat (e.g., areas surrounding the breeding colonies) should be surveyed to evaluate their importance as foraging grounds for the species, and could presumably be integrated into a MPAs network linking the distant foraging grounds identified in the Iberian continental shelf (Roberts et al. 2001).

In addition to the core foraging grounds, other more diffuse protective measures could be implemented within a larger buffer region delineated by the range of the species. Namely, a compulsory fishery observer program to record potential bycatch in longline fisheries, and the study of potential impacts of wind farms would apply to the continental shelf region, especially in the vicinity of the Ebro Delta. Currently, a marine wind farm is planned just a few kilometres off the Ebro Delta.

While we acknowledge that effective conservation of the Balearic shearwater may require the implementation of protective measures throughout the life cycle of the species, we recognize that MPAs identified in this study (restricted to the chick-rearing period) may not be feasible or effective during the post-breeding and juvenile dispersal stages. Thus, we advocate the judicious use of marine zoning strategies for MPAs

identification outside the breeding season in conjunction with other non site-specific management measures (e.g., bycatch mitigation and oil spill prevention programs), when the species disperses into the Western Mediterranean and the North Atlantic. Ultimately, non site-specific management measures should always complement site-specific (MPAs) measures, even when a well designed and thorough MPAs network is implemented.

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References

- Abelló P, Oro D (1998) Offshore distribution of seabirds in the Northwestern Mediterranean in June 1995. *Colonial Waterbirds* 21:422-426
- Abelló P, Guerao G (1999) Temporal variability in the vertical and mesoscale spatial distribution of crab Megalopae (Crustacea: Decapoda) in the Northwestern Mediterranean. *Estuarine, Coastal and Shelf Science* 49:129-139
- Abelló P, Arcos JM, Gil de Sola L (2003) Geographical patterns of seabird attendance to a research trawler along the Iberian Mediterranean coast. *Scientia Marina* 67S2:69-75

- Agostini VN, Bakun A (2002) 'Ocean triads' in the Mediterranean Sea: physical mechanisms potentially structuring reproductive habitat suitability (with example application to European anchovy, *Engraulis encrasicolus*). *Fisheries Oceanography* 11:129-142
- Anderson DJ, Huyvaert KP, Wood DR, Gillikin CL, Frost BJ, Mouritsen H (2003) At-sea distribution of Waved Albatrosses and the Galápagos Marine Reserve. *Biological Conservation* 110:367-373
- Arcos JM, Oro D (2002) Significance of fisheries discards for a threatened Mediterranean seabird, the Balearic shearwater *Puffinus mauretanicus*. *Marine Ecology Progress Series* 239:209-220
- Arcos JM, Oro D (2004) Pardela Balear, *Puffinus mauretanicus*. In: Madroño A, González C Atienza JC (Eds) Libro Rojo de las Aves de España. Dirección General para la Biodiversidad-SEO/BirdLife, Madrid, Spain, pp 46-50
- Arcos JM, Massutí E, Abelló P, Oro D (2000) Fish associated with floating drifting objects as a feeding resource for Balearic Shearwaters *Puffinus mauretanicus* during the breeding season. *Ornis Fennica* 77:177-182
- Arcos JM, Louzao M, Oro D (2006) Fishery Ecosystem Impacts and Management in the Mediterranean: Seabirds Point of View. In: Nielsen J, Dodson J, Friedland K, Hamon T, Hughes N, Musick J, Verspoor E (Eds) Proceedings of the Fourth World Fisheries Congress: Reconciling Fisheries with Conservation. American Fisheries Society, Symposium 49, Bethesda, Maryland, in press.
- Arnau P, Liqueste C, Canals M (2004) River mouth plume events and their dispersal in the Northwestern Mediterranean Sea. *Oceanography* 17:23-31
- Baillie JEM, Hilton-Taylor C, Stuart SN (Eds) (2004) IUCN Red List of Threatened Species. A Global Species Assessment. IUCN, Gland, Switzerland and Cambridge, UK
- Bearzi G, Reeves RR, Notarbartolo-Di-Sciara G, Politi E, Cañadas A, Frantzis A, Mussi B (2003) Ecology, status and conservation of short-beaked common dolphins *Delphinus delphis* in the Mediterranean Sea. *Mammal Review* 33:224-252
- Begg GS, Reid JB (1997) Spatial variation in seabird density at a shallow sea tidal mixing front in the Irish Sea. *ICES Journal of Marine Science* 54:552-565
- Bertrand JA, Gil de Sola L, Papaconstantinou C, Relini G, Souplet A (2002) The general specifications of the MEDITS surveys. *Scientia Marina* 66S2:9-17

- BirdLife International (2004) Tracking ocean wanderers: the global distribution of albatrosses and petrels. Results from the Global Procellariiform Tracking Workshop, 1-5 September, 2003, Gordon's Bay, South Africa. Cambridge, UK: BirdLife International
- Boyce MS, Vernier PR, Nielsen SE, Schmiegelow FKA (2002) Evaluating resource selection functions. *Ecological Modeling* 157:281-300
- Camphuysen CJ, Garthe S (1997) An evaluation of the distribution and scavenging habits of northern fulmars (*Fulmarus glacialis*) in the North Sea. *ICES Journal of Marine Science* 54:654-683
- Cañadas A, Sagarminaga R, De Stephanis R, Urquiola E, Hammond PS (2005) Habitat preference modeling as a conservation tool proposals for marine protected areas for cetaceans in southern Spanish waters. *Aquatic Conservation* 15: 495-521
- Cardona L, Revelles M, Carreras C, San Felix M, Gazo M, Aguilar A (2005) Western Mediterranean immature loggerhead turtles: habitat use in spring and summer assessed through satellite tracking and aerial surveys. *Marine Biology* 147:583-591
- Crawley MJ (1993) *GLIM for Ecologists*. Blackwell Scientific, Oxford, UK
- Demestre M, Lleonart J, Martín P, Recasens L, Sánchez P (1988) La pesca en Cataluña. *FAO Rapport Pêches* 395:101-103
- De Segura AG, Tomas J, Pedraza SN, Crespo EA, Raga JA (2003) Preliminary patterns of distribution and abundance of loggerhead sea turtles, *Caretta caretta*, around Columbretes Islands Marine Reserve, Spanish Mediterranean. *Marine Biology* 143:817-823
- EEC (1979) Council Directive 79/409/EEC of 2 April 1979 on the conservation of wild birds. In: *Official Journal L* 103, 25/04/1979. European Economic Community, Brussels, Belgium, p. 1
- EEC (1992) Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. In: *Official Journal L* 206 , 22/07/1992. European Economic Community, Brussels, Belgium, p. 7
- Estrada M (1996) Primary production in the northwestern Mediterranean. *Scientia Marina* 60:55-64
- Font J, Salat J, Tintoré J (1988) Permanent features in the circulation of the Catalan Sea. *Oceanologia Acta* 9:51-57

- Forney KA (2000) Environmental models of cetacean abundance: reducing uncertainty in population trends. *Conservation Biology* 14:1271-1286
- Franks PJS (1992) Swim or sink: accumulation of biomass at fronts. *Marine Ecology Progress Series* 82:1-12
- Gerber LR, Hyrenbach KD, Zacharias MA (2005) Do the largest reserves protect whales or whalers? *Science* 307:525-526
- Halpern BS (2003) The impact of marine reserves: Do reserves work and does reserve size matter? *Ecological Applications* 13:S117-S137
- Heinemann D (1981) A rangefinder for pelagic bird censusing. *Journal of Wildlife Management* 45:489-493
- Hooker SB, McClain CR (2000) The calibration and validation of SeaWiFS data. *Progress in Oceanography* 45:427-465
- Hooker SK, Whitehead H, Gowans S (1999) Marine protected area design and the spatial and temporal distribution of cetaceans in a submarine canyon. *Conservation Biology* 13:592-602
- Hooker SK, Gerber LR (2004) Marine reserves as a tool for ecosystem-based management: the potential importance of megafauna. *Bioscience* 54:27-39
- Houde E (Ed) (2001) *Marine Protected Areas: Tools for Sustaining Ocean Ecosystems*. National Academy Press, Washington DC, USA
- Huettmann F, Diamond AW (2000) Seabird migration in the Canadian northwest Atlantic Ocean: moulting locations and movement patterns of immature birds. *Canadian Journal of Zoology* 78:624-647
- Hunt GL Jr, Coyle KO, Hoffman S, Decker MB, Flint EN (1996) Foraging ecology of Short-tailed Shearwaters near the Pribilof Islands, Bering Sea. *Marine Ecology Progress Series* 141:1-11
- Hurlbert SH (1984) Pseudoreplication and the design of ecological field experiments. *Ecology* 54:187-211
- Hyrenbach KD, Forney KA, Dayton PK (2000) Marine protected areas and ocean basin management. *Aquatic Conservation* 10:437-458
- Hyrenbach KD, Fernández P, Anderson DJ (2002) Oceanographic habitats of two sympatric North Pacific albatrosses during the breeding season. *Marine Ecology Progress Series* 233:283-301
- Hyrenbach KD, Keiper C, Allen SG, Anderson DJ, Ainley DG (2006) Use of national marine sanctuaries by far-ranging predators: commuting flights to the

- California Current System by breeding Hawaiian albatrosses. *Fisheries Oceanography* 15: 95-103
- James MC, Ottensmeyer CA, Myers RA (2005) Identification of high-use habitat and threats to leatherback sea turtles in northern waters: new directions for conservation. *Ecology Letters* 8:195-201
- Kahru M (2000) Windows Image Manager - Image display and analysis program for Microsoft Windows with special features for satellite images (available at: <http://wimsoft.com>)
- Kahru M, Mitchell BG (2000) Influence of the 1997-98 El Niño on the surface chlorophyll in the California Current. *Geophysical Research Letters* 27:2937-2940
- LeFevre J (1986) Aspects of the biology of frontal systems. *Advances in Marine Biology* 23:163-299
- Lloret J, Lleonart J, Solé I, Fromentin J-M (2001) Fluctuations of landings and environmental conditions in the north-western Mediterranean Sea. *Fisheries Oceanography* 10:33-50
- Longhurst A (1998) *Ecological geography of the sea*. Academic Press, San Diego, CA
- Louzao M, Igual JM, McMinn M, Aguilar JA, Triay R, Oro D (2006) Small pelagic fish, trawling discards and breeding performance of the critically endangered Balearic Shearwater: improving conservation diagnosis. *Marine Ecology Progress Series* 318:247-254
- Manly BFJ (1994) *Multivariate Statistical Methods*. 2nd Edition. Chapman and Hall, New York, USA
- Millot C (1999) Circulation in the Western Mediterranean Sea. *Journal of Marine Systems* 20:423-442
- Millot C, Taupier-Letage I (2005) Circulation in the Mediterranean Sea. In: Saliot A (Ed) *The Mediterranean Sea: the Handbook of the Environmental Chemistry*, vol. 5 Water Pollution. Springer-Verlag Editor, pp. 1-30
- Nagao K, Ota S, Hirono J (1993) Regulation of the Japanese high seas drifnet fisheries. *International North Pacific Fisheries Commission Bulletin* 53:39-44
- NGDC (National Geophysical Data Center) (1998) Data Announcement 88-MGG-02, digital relief of the surface of the Earth. Boulder, Colorado, USA (www.ngdc.noaa.gov/mgg/global/seltopo.html)

- NOAA (2002) Taking of threatened or endangered species incidental to commercial fishing operations. Federal Register 67:59243-59245
- Norse EA, Crowder LB, Gjerde K, Hyrenbach KD, Roberts C, Soulé ME (2005) The potential for reserves as an ecosystem-based management tool in the open ocean. In: Norse EA, Crowder LB (Eds) Marine Conservation Biology. Island Press, Washington, USA, pp. 302-327
- Oro D, Aguilar JS, Igual JM, Louzao M (2004) Modelling demography and extinction risk in the endangered Balearic shearwater. Biological Conservation 116:93-102
- Palanques A, García-Ladona E, Gomis D, Martín J, Marcos M, Pascual A, Puig P, Gili J-M, Emelianov M, Monserrat S, Guillén J, Tintoré J, Segura M, Jordi A, Ruiz S, Basterretxea G, Font J, Blasco D, Pagès F (2005) General patterns of circulation, sediment fluxes and ecology of the Palamós (La Fonera) submarine canyon, northwestern Mediterranean. Progress in Oceanography 66:89-119
- Pearce J, Ferrier S (2000) Evaluating the predictive performance of habitat models developed using logistic regression. Ecological Modeling 133:225-245
- Pearson RG, Dawson TP, Liu C (2004) Modeling species distributions in Britain: a hierarchical integration of climate and land-cover data. Ecography 27:285-298
- Pikitch EK, Santora C, Babcock EA, Bakun A, Bonfil R, Conover DO, Dayton P, Doukakis P, Fluharty D, Heneman B, Houde ED, Link J, Livingston PA, Mangel M, McAllister MK, Pope J, Sainsbury KJ (2004) Ecosystem-based fishery management. Science 305:346-347
- Planes S, Galzin R, Garcia Rubies A, Goñi R, Harmelin J-G, Le Diréach L, Lenfant P, Quetglas A (2000) Effects of marine protected areas on recruitment processes with special reference to Mediterranean littoral ecosystems. Environmental Conservation 27:126-43
- Raymond B, Woehler EJ (2003) Predicting seabirds at sea in the Southern Indian Ocean. Marine Ecology Progress Series 263:275-285
- Roberts CM, Halpern B, Palumbi SR, Warner RR (2001) Designing marine reserve networks. Conservation Biology in Practice 2:10-17
- Ruiz A, Martí R (Eds) (2004) La Pardela Balear. SEO/BirdLife-Conselleria de Medi Ambient del Govern de les Illes Balears, Madrid, Spain
- Sabatés A (1996) Distribution pattern of larval fish populations in the Northwestern Mediterranean. Marine Ecology Progress Series 59:75-82

- Salat J (1996) Review of hydrographic environmental factors that may influence anchovy habitats in the northwestern Mediterranean. *Scientia Marina* 60S2:21-32
- Salat J, Garcia MA, Cruzado A, Palanques A, Arín L, Gomis D, Guillén J, de León A, Puigdefàbregas J, Sospedra J, Velásquez ZR (2002) Seasonal changes of water mass structure and shelf slope exchanges at the Ebro Shelf (NW Mediterranean) *Continental Shelf Research* 22:327-348
- SAS Institute (2000) SAS/STAT® software: User's guide. Cary, North Carolina, USA
- Sawada M (1999) Rookcase: an excel 97/2000 visual basic (VB) add-in for exploring global and local spatial autocorrelation. *Bulletin of the Ecological Society of America* 80:231-234
- Schneider DC (1990) Spatial autocorrelation in marine birds. *Polar Research* 8:89-97
- Schneider DC (1993) The role of fluid dynamics in the ecology of marine birds. *Oceanography and Marine Biology: An Annual Review* 29:487-521
- Schneider DC (1997) Habitat selection by marine birds in relation to water depth. *Ibis* 139:175-178
- Skov H, Durinck J (2001) Seabird attraction to fishing vessels is a local process. *Marine Ecology Progress Series* 214:289-298
- Sokal RR, Oden NL (1978) Spatial autocorrelation in biology. I. Methodology. *Biological Journal of the Linnean Society* 10:199-228
- Sokal RR, Oden NL, Thomson BA (1998) Local spatial autocorrelation in biological variables. *Biological Journal of the Linnean Society* 65:41-62
- Spear LB, Balance LT, Ainley DG (2001) Response of seabirds to thermal gradients in the tropical Pacific: the thermocline versus the Equatorial Front. *Marine Ecology Progress Series* 219:275-289
- SPEA-SEO/BirdLife (2005) Implementing N2000 in the marine environment. Marine IBAs: Lisbon-Vilanova conclusions (Available at <http://www.seo.org/media/docs/Marine%20IBAs%20Lisbon-Vilanova%20conclusions.pdf>)
- Swests KA (1988) Measuring the accuracy of diagnostic systems. *Science* 240:1285-1293
- Tasker ML, Jones PH, Dixon T, Blake BF (1984) Counting seabirds at sea from ships: a review of methods employed and a suggestion for a standardized approach. *Auk* 101:567-577

- Thuiller W, Vayreda J, Pino J, Sabate S, Lavorel S, Gracia C (2003) Large-scale environmental correlates of forest tree distributions in Catalonia (NE Spain). *Global Ecology and Biogeography* 12:313-325
- Van Erkel AR, Pattynama PMT (1998) Receiver operating characteristic (ROC) analysis: basic principles and applications in radiology. *European Journal of Radiology* 27:88-94
- Watkins MW (2000) An EXCEL program for calculating and graphing the Receiver Operating Characteristic (ROC). Ed and Psych Associates. State College, Pennsylvania, USA
- Williams G, Pullan D, Dickie I, Huggett D, Mitchell H (2005) The European Birds Directive - safeguarding special places for people and wildlife. RSPB, Sandy
- Zar JH (1999) *Biostatistical Analysis*. 4th Edition. Prentice-Hall, New Jersey, NJ

Chapter 6



Seabird attendance to the trawling fishery at the Balearic archipelago (western Mediterranean)

Maite Louzao, José Manuel Arcos & Daniel Oro

6.1 Abstract

Bottom trawl fishing provides substantial amounts of demersal discards (a naturally unavailable trophic resource) to seabirds, affecting their life-history traits and population dynamics, as well as community structure. This study reports on seabird attendance to a bottom trawling fishery at the Balearic archipelago (western Mediterranean) throughout the year. Particular attention is devoted to species of conservation concern. We characterised seabird distribution patterns and community structure in relation to fishery-related variables. Globally, the spatio-temporal coupling of seabird abundance was a result of a complex interaction between cross-correlated fishery-related variables. Analyses of the spatio-temporal patterns were conducted for each breeding species separately, showing a significant influence of the season (relative to their annual cycle) in the number of seabirds attending trawlers. The most abundant breeding species, the yellow-legged gull *Larus michaellis* and Cory's shearwater

Calonectris diomedea, were also the most common species behind trawlers. The latter attended vessels in higher numbers than expected from its local population densities. On the contrary, the remaining breeding species occurred in lower numbers than expected, which suggest that discards were of relative little importance to them. The core area of the discarding activity took place in the continental shelf off the south-west coast of Mallorca influencing 12% of the total breeding seabird numbers. Regarding waste availability, overall discards corresponded to 74% of landings (range: 9 – 600%), though intrinsic and habitat characteristics of each fishing stratum influenced this ratio (being the slope stratum the most selective). This allowed estimating an approximate average of 1050 tons of discards annually in Mallorca. Both discard composition and landings were characterised by a major proportion of fish followed by crustacean and mollusc (ca. 80%, 15%, and 5%, respectively). Ultimately, an ecosystem-based approach is necessary in order to integrate the seabird community in the western Mediterranean fisheries management, especially urgent for the critically endangered Balearic shearwater *Puffinus mauretanicus*.

6.2 Introduction

Marine top predators play an important role on the ecosystem functioning and are especially affected by environmental changes (including human-induced perturbations such as fishing) due to their high trophic position in the food web (Furness & Camphuysen 1997, Heppel et al. 2005, Arcos et al. 2006). Some predators, such as seabirds, interact with commercial fisheries in different ways: in some cases, fisheries promote high mortality driving some species to near extinction (see reviews in Brothers et al. 1999, Lewison et al. 2004, Myers & Worm 2003, 2005), while in other cases fisheries deplete fish stocks when target the same or interconnected prey than seabirds, and so reduce their availability (Furness 2003). However, some fishing activities (e.g. bottom trawling) provide a naturally unavailable trophic resource to scavenging seabirds: demersal species in the form of offal and discards (Tasker et al. 2000, Arcos et al. 2006). Availability of such fishery waste can actually affect life-history traits of seabirds and their population dynamics, as well as community structure (Oro 1999, Votier et al. 2004). Indeed, discards provide an important food supply to scavenging species which could lead to improve breeding performance (e.g. feeding rates to chicks, clutch and egg size) and increased survival of seabirds (e.g. Oro et al.

1995, Oro & Furness 2002, Oro et al. 2004a). Consequently, this human-induced food resource could potentially favour the more generalist species, sustaining their increasing population numbers (Oro 1999, Furness 2003).

During the 1990s, ca. 27 million tons of fish were discarded worldwide each year (Alverson et al. 1998). In the European Union, incoming fisheries policies are directed at reducing discards and bycatch (COM 2002), and in turn decreasing the availability of this trophic resource (in the short term) for scavenging species. Thus, opportunistic seabirds would be forced to seek for alternative foraging resources (Stenhouse & Montevecchi 1999). Discards shortage could adversely affect seabird breeding performance and population size of scavenging species, as well as to the entire seabird community (Reeves & Furness 2002). This fishing regulation could also promote intra and inter-specific competition as well as predation on more specialised and vulnerable taxa (Oro 1996, Stenhouse & Montevecchi 1999, Furness 2003, Votier et al. 2004). Particularly sensitive are those species with low foraging plasticity, i.e. species with energetically expensive foraging methods, restricted to forage close to the colony or unable to dive and allocate food resources (Furness & Ainley 1984, Arcos et al. 2001).

In the Mediterranean Sea, fishing activity is characterised by highly diversified catches and the absence of large single stocks, especially in the demersal fisheries (Farrugio et al. 1993, Lleonart & Maynou 2003). In the western Mediterranean, most demersal stocks are fully exploited or overexploited, while some pelagic stocks also show overexploitation trends (Farrugio et al. 1993, Coll et al. 2006). Fishing discards are generated mainly in trawling and, to a lesser extent, purse-seine fisheries (e.g. Carbonell et al. 1998, Arcos & Oro 2002a, Arcos & Oro 2002b, Tudela 2004) totalling ca. 565.000 tons per year in the whole Mediterranean and Black Sea (Alverson 1998). Considerable attention has been devoted to the utilisation of fishery waste by seabirds, mostly during the breeding season in the Iberian continental shelf (Arcos 2001, Martínez-Abraín et al. 2002), although studies are scarce in other areas (e.g. the Balearic archipelago - but Oro & Ruiz 1997). Also, investigations outside the breeding season should be conducted (but Arcos 2001) since the effects of resource availability are not as evident as in the breeding period, but still can be crucial for survival (Huppopp & Wurm 2000, Martínez-Abraín et al. 2002, Grosbois & Thompson 2005).

This study reports on seabird attendance to a trawling fishery at the Balearic archipelago throughout the year, paying particular attention to species of conservation concern. We characterised the distribution patterns and community structure of

seabirds from two different approaches: (1) from a global approach we modelled, for the first time, seabird abundance and community diversity in association with trawlers in relation to fishery-related variables; and (2) a species-specific approach allowed exploring the spatio-temporal patterns of attendance of breeding species. We also assessed (3) trawling-related variables that could influence seabird attendance, such as waste availability and specific composition of discards available for seabirds; and (4) the geographic area influenced by the discarding activity of the main trawling harbour of the archipelago, estimated by Kernel analysis, novel in studies of seabird attendance to trawling fishery.

6.3 Material and Methods

Study area, trawling fishery and seabird community

The Balearic archipelago is characterised by a narrow continental shelf, with absence of submarine canyons, and lack of sediment input from the shelf (i.e. absence of river runoffs). Water masses circulating through its channels shape its high hydrographical variability (e.g. Pinot et al. 2002). Since the archipelago is separated from the Iberian Peninsula by large geographical barriers (depths between 800 and 2000 m), it can be considered as an isolated demersal ecosystem (Massutí & Reñones 2005). The trawling fishery is one of the most important fisheries, in catches and economic terms, in the western Mediterranean (Carbonell et al. 1998, Lleonart & Maynou 2003). In 2003, the semi-industrial trawling fleet of the Balearic Islands comprised 55 vessels at 16 harbours, with a decreasing trend in recent years. Trawlers operate at different fishing strata depending on the target species, between 50 and 800 m depth: striped red *Mullus surmuletus* and red mullets *M. barbatus* from coastal down to 100 m (hereafter coastal fishing stratum); European hake *Merluccius merluccius* and slope-water rose shrimp *Parapenaeus longirostris* in the shelf, between 150 and 350 m (hereafter shelf fishing stratum); and red shrimp (*Aristeus antennatus*) in the slope, from 350 to 800 m (hereafter slope fishing stratum) (e.g. Carbonell et al. 2003, Guijarro & Massutí 2006). The trawling fleet operates only on working days during 12 h (from 5am to 5pm). In this area no trawling moratorium has been established, in contrast to other areas off the Mediterranean Iberian Peninsula (e.g. Oro 1999).

The level of endemism of the Mediterranean ecosystem is high, including seabirds, thus being of special conservation concern (Bianchi & Morri 2000). The seabird breeding community presents small population sizes, and it is especially diverse at the Balearic archipelago, where six species breed regularly (Zotier et al. 1999): three petrel species (the Balearic shearwater *Puffinus mauretanicus*, endemic to the archipelago, the Cory's shearwater *Calonectris diomedea* and the European storm-petrel *Hydrobates pelagicus*), the Mediterranean shag *Phalacrocorax aristotelis desmarestii*, and two gull species (Audouin's gull *Larus audouinii* and the yellow-legged gull *L. michaellis*) (see Table 6.1) (Aguilar 1991, Viada 2006). Total numbers are estimated at ca. 31000 breeding pairs (data for 1991). Non breeding scavenging species also occur regularly at the Balearic archipelago, in small numbers, mostly during their non-breeding period (e.g the Northern gannet *Morus bassanus*, and the great skua *Stercorarius skua*; Rebassa 2003).

Data collection

Data were collected on board 8 trawlers belonging to 4 different ports from Mallorca and 2 ports from Eivissa and Formentera (there are a total of 3 ports in both islands). In total, we conducted 31 one-day cruises recording 46 fishing operations, from April 2002 to September 2003 (Figure 6.1). We counted the seabirds attending the trawler during the discarding process, starting when the net was on the surface and stopping when all discards were thrown overboard. Also, the number of trawlers operating at the beginning of the discarding process in the fishing area was recorded. For each discarding process, stern counts were carried out every 15 min, from the position that offered the best view within 360° (Camphuysen et al. 1995), recording the species, number and age (adults and immatures) of seabirds following the trawler. Since consecutive censuses of seabirds following vessels, as well as their geographic positions may not be statistically independent, we considered the maximum number per species and the average position of each discarding process as the representative count and position for each process, respectively (cf. Oro & Ruiz 1997). Depending on the type of fishing stratum, vessels made only one (mainly in the slope stratum) or a varying number of consecutive hauls per day (up to three in the case of coastal stratum).

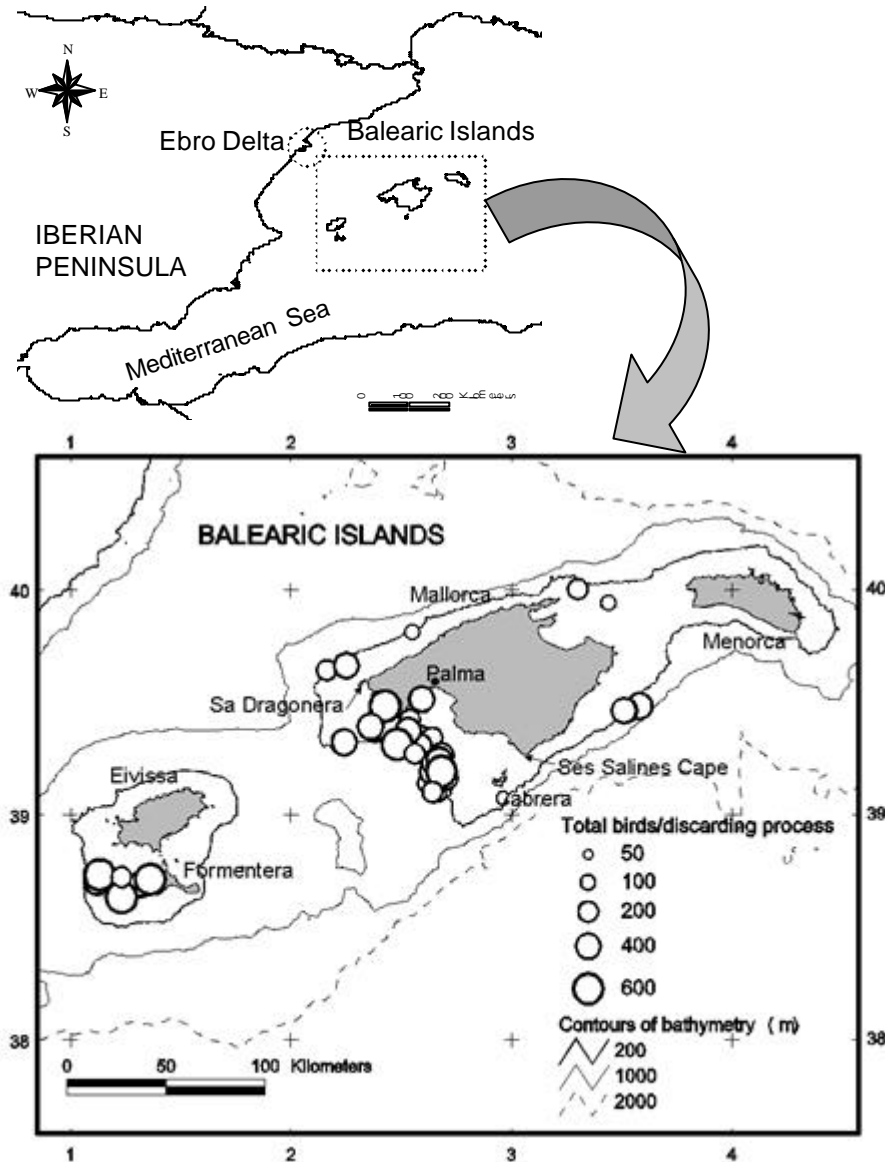


Figure 6.1 Map of the study area showing total number of birds per discarding process. Geographic location are in °N and °E of latitude and longitude, respectively. Circle sizes are proportional to seabird numbers. Geographic references mentioned in the text are indicated.

For each trawler, we noted the horsepower and length. When possible, we recorded the amount of discards and landings (i.e. marketable catches) for each fishing operation. Since discards were thrown directly overboard, most frequently using plastic shovels but also by hand, we counted the number of these actions, and weighted the content of a representative number of them (usually 5-10) to obtain an average weight, which allowed us estimating a total weight of discards (cf. Oro & Ruiz 1997, Arcos 2001). Landings for each fishing trip were estimated consulting the fishermen guild statistics. We calculated a ratio of discards vs. landings (discard ratio, $DR = \text{discards} / \text{landings}$) (cf. Arcos 2001). Total annual landings from Mallorca, for

the study period (2002 and 2003), were provided by the Spanish Institute of Oceanography. In addition, a sub-sample of the discard was separated for later examination in the laboratory. Taxonomic groups were identified, at the species level when possible, and weighted. In the case of fish species also length was measured, to the nearest millimetre.

Statistical analysis

For each haul, the descriptive statistics of seabird counts were given (mean, median and range) as well as the numerical percentage of each species (%N), and the percentage of occurrence (%P; proportion of hauls where the species was recorded). For the fishery-related variables, we first explored the normality of the data using the Shapiro-Wilk test. When this assumption was not met, either data were transformed (by $\ln(x+1)$) or non-parametric statistics were used.

Modelling seabird attendance to a trawling fishery: a global approach. We analysed seabird attendance to the Balearic trawling fishery globally, taking into account both total seabird abundance and community diversity, in order to identify the most significant explicative variables. Community diversity was estimated using the Shannon-Weaver index, which takes into account both the number of species and the relative abundance of individuals of each species, calculated as follows:

$$H' = - \sum_{i=1}^k p_i \ln p_i$$

where H' = Shannon-Weaver index, k = number of species, and p_i = proportion of birds of species i relative to the total abundance (Krebs 2001).

We gathered sufficient information on 31 discarding processes for analysing 7 explicative variables: horse power, fishing stratum, discarding process duration, amount of discards, number of trawlers at the beginning of the discarding activity in the fishing area, depth and nearest distance to the shoreline. Fishery-related variables such as horsepower was selected as explicative variable since more powerful boats were expected to catch more fish and provide larger amounts of discards (during a longer period), and ultimately enhance seabird attendance. Also, the same result was expected when an increased number of trawlers occurred simultaneously in the fishing area. Fishing stratum, defined on the basis of trawling depth (associated to target species), could influence seabird distribution as bathymetry does (Schneider 1997, Yen

et al. 2004), as well as the distance to the shoreline (Louzao et al. 2006a). Depth was obtained from NOAA's ETOPO 5-minute (spatial resolution of 7.5 – 8.2 km at 35° N and 42° N latitude, respectively) gridded elevation dataset (NGDC 1998), and extracted using the WIM 6.26 software (Kahru 2000). We also expected seasonal differences on seabird attendance since some breeding species largely leave their breeding sites after reproduction (the three petrel species and Audouin's gull), whereas others (the Mediterranean shag and the yellow-legged gull) tend to stay at the archipelago. For our study, we divided the annual cycle of breeding seabirds in four different periods: December-February, March-May, June-August and September-November (winter, spring, summer and autumn hereafter, respectively). These periods roughly match different stages of the annual cycle of seabirds, details depending on the species.

After exploratory analyses, we ln-transformed the horse power, discarding duration, discards, number of trawlers at the beginning of the discarding activity, and depth. Since all these variables were cross-correlated (Table 6.2), we used generalized linear models (GENMOD procedure in SAS Institute 2000) to assess the relative influence of each variable on total seabird abundance and community diversity. In the case of seabird counts, we built the model assuming a Poisson distribution and using the log-link function assembling the best-fit model iteratively using a backwards step-wise procedure by retaining the variables with the highest explanatory power one at a time (variables were retained whenever $\alpha < 0.05$). For the community diversity, we repeated the same procedure implementing a normal distribution and the identity-link function. The interpretation of these results is based on the sign of the estimated response coefficients, where a positive and negative signs are indicative of a greater and a smaller probability of recording higher or lower seabird abundance/diversity in a given discarding process. Categorical variables were calculated relative to the last category (indicated by null values of the estimated parameter and standard error).

Seabird breeding community: a species-specific approach. We explored seasonal and fishing stratum effects on each breeding species abundance, as well as geographic patterns of trawling attendance (using ArcView 3.2.). In addition, we assessed whether breeding species were making a greater use of discards than expected from their numbers of breeding pairs during spring off south-west Mallorca (main area surveyed, see Figure 6.1). Numbers of breeding pairs were estimated in this area defined by Sa Dragonera Island as the northern limit and Ses Salines Cape - Cabrera Island as the southern limit (see Table 6.1 and Figure 6.1). Assuming that all breeding species were

equally likely to follow a trawler, we tested the simple null hypothesis that the six breeding species followed trawlers proportionally to their densities in the area (Oro & Ruiz 1997).

Table 6.1 Breeding pairs of seabirds estimated at the south-west coast of Mallorca, defined by Sa Dragonera Island as the northern limit and Ses Salines Cape - Cabrera Island as the southern limit (see Figure 6.1). The percentage of breeding pairs in south-west Mallorca relative to the total breeding population at the archipelago for each species in 1991 is also shown (see Material and Methods for details) (Aguilar 1991, Viada 2006).

Species	Breeding pairs (SW Mallorca)		% relative to total breeding population (1991)
	1999-2003	1991	
Balearic shearwater	530	428	13.0
Cory's shearwater	635	686	6.2
European storm-petrel	419	60	2.1
Mediterranean shag	230	310	34.8
Audouin's gull	458	118	15.4
Yellow-legged gull	5654	> 2000	16.5

Area of discarding activity. We identified the geographic area where discarding activity occurred throughout the year. Cruises were mainly conducted on board vessels from Palma harbour (the main trawling harbour of the archipelago), thus we centred this analysis in hauls performed by this fleet ($n = 33$) (Figure 6.1). We used a non-parametric density function, the fixed kernel utilisation distribution, which draws contours about areas of equal density from the observed data (i.e. average positions of discarding processes) (Worton 1989, Selkirk & Bishop 2002). Each contour represents a specific probability of finding a particle within that area at a particular time (Anderson 1982). A grid coverage was built using the default least squares cross validation (LSCV) representing the 50% (core areas of influence) and 95% (overall range of influence) of the probability of occurring a discarding process during the whole study period. The 50% and 95% estimates eliminate 50% and 5% of the positions, which are considered outliers. For this purpose, we used the Animal Movement Analysis extension on ArcView 3.2 (Hooge & Eichenlaub 1997).

6.4 Results

The location of the 46 hauls recorded during the study is shown in Figure 6.1. Overall, discarding processes occurred over the continental shelf (median depth 127 m,

range 4 – 259; n = 45) close to the coast (median distance to the shoreline 15.3 km, range 2.0 – 28.7; n = 45), which lasted a median of 75 minutes (range 30-195; n = 46). The surveyed trawling fleet was characterised by a median horse power and vessel length of 170 (range: 75 – 430) and 17.5 m (range: 13.2 – 21.7), respectively. We recorded a total number of 11109 birds attending vessels with a median of 227 individuals (range: 31-530; n = 46) per haul (Table 6.3). Regarding seasons, 10.9%, 41.3%, 34.8% and 13.0% of the fishing operations were recorded in winter, spring, summer and autumn, respectively. By fishing stratum, coastal hauls were more frequent (52.2%) than shelf and slope hauls (each representing 23.9% of the total).

Characterising the trawling fishery: waste availability and discard composition

The cross-correlation showed that vessel power was positively correlated with discarding process duration. At the same time, longer discarding duration took place at higher depth and distance to the shoreline (positive correlation) (Table 6.2). At the slope fishing stratum we recorded the highest number of trawlers operating at the beginning of the discarding process (positive correlation).

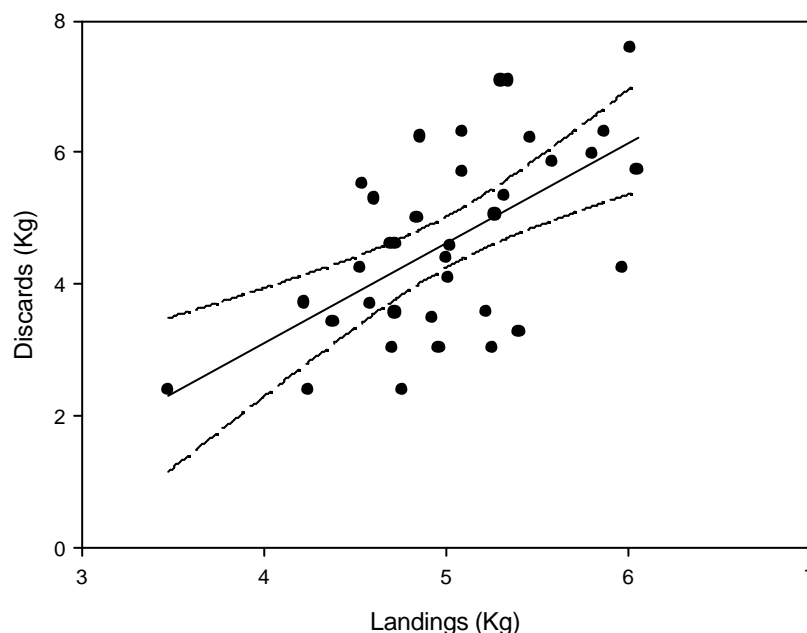


Figure 6.2 Linear regression and 95% confidence intervals of fish discarded vs. fish landed (both variables $\ln(x+1)$ transformed). Pearson correlation coefficient between both \ln -transformed variables was significant ($r = 0.592$, $n = 36$, $p < 0.001$).

Table 6.2 Bivariate Spearman rank P-values and correlations between vessel-related variables analysed in the GLMs. NS = non-significant.

	Season	Fishing stratum	Horsepower	Discarding duration	Discards	Trawlers	Depth	Distance shoreline
Season	1.000	NS	NS	NS	NS	NS	NS	NS
Fishing stratum	0.133	1.000	<0.001	NS	NS	<0.01	NS	NS
Horsepower	0.155	0.598	1.000	<0.001	NS	NS	NS	NS
Discarding duration	0.168	0.014	0.506	1.000	NS	NS	<0.05	NS
Discards	-0.042	-0.240	0.060	0.045	1.000	<0.001	NS	NS
Trawlers	-0.135	0.457	-0.090	-0.165	-0.540	1.000	NS	NS
Depth	0.105	0.031	0.300	0.438	0.006	-0.104	1.000	<0.001
Distance shoreline	0.134	0.047	0.248	0.345	-0.057	-0.055	0.565	1.000

Table 6.3 Number of fishing operations (n), discarding duration (median and range in minutes) and total number of birds attending discarding processes are given per season and fishing stratum. Also total numbers for both categories are given. Note that no discarding process was recorded in the coastal stratum in winter.

Season	General information	Fishing stratum			Total by season
		Coastal	Shelf	Slope	
Winter	n	-	3	2	5
	Discarding duration	-	90.0 (30 - 90)	112.5 (105 - 120)	90.0 (30 - 120)
	Total number of seabirds	-	290.0 (208 - 453)	184.5 (35 - 334)	290.0 (35 - 453)
Spring	n	14	3	2	19
	Discarding duration	60.0 (30 - 135)	120.0 (60 - 120)	112.5 (45 - 180)	60.0 (30 - 180)
	Total number of seabirds	261.5 (128 - 530)	152.0 (120 - 495)	205.0 (178 - 232)	248.0 (120 - 530)
Summer	n	9	3	4	16
	Discarding duration	90.0 (45 - 195)	75.0 (60 - 165)	75.0 (45 - 120)	75.0 (45 - 195)
	Total number of seabirds	136.0 (31 - 466)	239.0 (215 - 322)	114.5 (90 - 270)	161.0 (31 - 466)
Autumn	n	1	2	3	6
	Discarding duration	105	90.0 (45 - 135)	105.0 (90 - 165)	105.0 (45 - 165)
	Total number of seabirds	411	260.5 (180 - 341)	131.0 (63 - 160)	170.0 (63 - 411)
Total by stratum	n	24	11	11	46
	Discarding duration	75.0 (30 - 195)	90.0 (30 - 165)	105.0 (45 - 180)	75.0 (30 - 195)
	Total number of seabirds	252.5 (31 - 530)	239.0 (120 - 495)	131.0 (35 - 334)	227.0 (31 - 530)

Table 6.4 Median (and range) of landings (kg), fish discards (kg) and discard ratio (DR = discards/ landings) by season and fishing stratum.

Season	Fishing stratum				Total by season
	General information	Coastal	Shelf	Slope	
Winter	n	-	-	1	1
	Discards	-	-	97.4	97.4
	Landings	-	-	150.0	150.0
	DR	-	-	0.65	0.65
Spring	n	13	1	2	16
	Discards	70.0 (10.0 - 2000.0)	1200.0	52.2 (34.0 - 70.0)	70.0 (10.0 - 2000.0)
	Landings	149.0 (31.1 - 406.0)	207.5	101.5 (92.0 - 111.0)	138.2 (31.1 - 406.0)
	DR	0.40 (0.11 - 6.04)	5.78	0.54 (0.31 - 0.76)	0.58 (0.11 - 6.04)
Summer	n	9	3	4	16
	Discards	210.0 (20.0 - 550.0)	313.6 (200.0 - 550.0)	26.1 (10.0 - 40.0)	178.1 (10.0 - 550.0)
	Landings	193.7 (92.5 - 354.5)	162.0 (98.0 - 424.5)	112.4 (67.0 - 136.0)	144.3 (67.0 - 424.5)
	DR	1.19 (0.14 - 2.70)	2.04 (0.74 - 3.39)	0.21 (0.09 - 0.60)	0.92 (0.09 - 3.39)
Autumn	n	1	1	1	3
	Discards	35.0	300.0	39.5	39.5 (35.0 - 300.0)
	Landings	184.0	161.0	97.0	161.0 (97.0 - 184.0)
	DR	0.19	1.86	0.41	0.41 (0.19 - 1.86)
Total by stratum	N	23	5	8	36
	Discards	100.0 (10.0 - 2000.0)	313.6 (200.0 - 1200.0)	37.0 (10.0 - 97.4)	98.7 (10.0 - 2000.0)
	Landings	184.0 (31.0 - 406.0)	162.0 (98.0 - 425.0)	109.9 (67.0 - 150.0)	148.1 (31.0 - 425.0)
	DR	0.90 (0.11 - 6.04)	2.04 (0.74 - 5.78)	0.36 (0.09 - 0.76)	0.74 (0.09 - 6.00)

Trawlers generated a median of 99 kg of discards (range: 10 - 2000) and 148 kg of landings (range: 31 - 425) per fishing operation (Table 6.4). Both ln-transformed variables were significantly correlated (Pearson correlation coefficient, $r = 0.592$, $n = 36$, $p < 0.001$) (Figure 6.2). The overall median of DR was estimated in 0.74 (range: 0.09 - 6.00). Discards, landings and DR differed by season and fishing stratum (Table 6.4). However, we only found a significant fishing stratum effect in these variables (ANOVA for ln-transformed discards $F_{2,36} = 5.88$, $p = 0.007$; ANOVA for ln-transformed landings $F_{2,44} = 4.18$, $p = 0.022$; Kruskal-Wallis test for DR $H_{2,36} = 7.83$, $p = 0.02$). In addition, contrast analyses showed that the shelf stratum generated a significantly

greater amount of landings and discards in relation to the slope stratum (Table 6.4). The latter was more selective, discarding in overall 36% of landings (range: 9 - 76%), whereas the shelf stratum discarded approximately twice the amount of landings (range: 74 - 578%) (Table 6.4).

From the total landings recorded (8104 kg), fish, crustaceans, and molluscs represented by mass 79.6%, 15.3% and 5.1%, respectively. This was similar to the percentages obtained from the discard sub-sample (70.9% fish, 5.9% crustaceans, 1.5% molluscs and 1.1% echinoderms, out of 123.3 kg analysed). The remainder 20.6% corresponded to unidentifiable organic matter, inorganic waste and unidentifiable waste. Discard composition included 66 taxonomic groups of fish (Appendix 1). For the study period, approximately a total of 1419 tons were landed annually in Mallorca. Applying the global median of DR (0.74, range: 0.09 - 6.00) to the approximate total annual landings, an overall of 1050 tons (range: 127 - 8515) were discarded (Table 6.4), from which 744, 62, and 16 tons corresponded to fishes, crustaceans and molluscs, respectively (based on the proportion of each taxonomic group in the discard sub-sample).

Table 6.5 Results of the generalized linear model (GLM) analysis of the total seabirds and community diversity recorded during the discarding processes. Only significant variables are shown. The effect of categorical variables (e.g. season and fishing stratum) was calculated relative to the last category (see the null value of winter and slope fishing stratum). S.E.: standard error, D.F.: degrees of freedom.

Variable		Estimate parameter	S.E.	D.F.	χ^2	Significance
<i>Total seabirds</i>						
Season	Winter	0.709	0.075	3	382.56	< 0.001
	Spring	0.240	0.047			
	Summer	-0.235	0.053			
	Autumn	0.000	0.000			
Fishing stratum	Coastal	1.095	0.058	2	514.55	< 0.001
	Shelf	1.098	0.052			
	Slope	0.000	0.000			
Horsepower		0.607	0.048	1	156.11	< 0.001
Discarding duration		-0.121	0.035	1	11.89	< 0.001
Trawlers		0.296	0.029	1	107.60	< 0.001
Distance to shoreline		-0.031	0.002	1	159.69	< 0.001
<i>Community diversity</i>						
Discards		-0.732	0.180	1	13.28	< 0.001
Trawlers		-0.926	0.385	1	5.30	0.021
Distance to shoreline		0.095	0.154	1	5.32	0.021

Factors influencing seabird attendance and community diversity

We identified six significant variables to explain seabird attendance: season, fishing stratum, horse power, discarding duration, number of trawlers at the beginning of the discarding process in the fishing area and distance to the nearest shoreline (Table 6.5). The probability of recording higher abundance of seabirds in a discarding process was higher in winter, at both the coastal and shelf fishing strata, at short distances from the shoreline, and when discarding duration was shorter (Table 6.3). Vessels with higher horsepower attracted more seabirds, as well as the number of trawlers presented in the fishing area. Summer was the period with lower seabird abundances.

Three explanatory variables were significantly associated with community diversity: amount of discards, trawlers at the beginning of the discarding process and distance to shoreline (Table 6.5). Diversity was higher when both the amount of discards and the number of trawlers at the beginning of the discarding process decreased, at greater distances to the shoreline.

Spatio-temporal patterns of breeding community

The Yellow-legged gull (77%) and Cory's shearwater (18%) were the most abundant species following trawlers (Table 6.6), as well as the most common breeding species (Table 6.1). The six breeding species occurred in more than 25% of the hauls (Table 6.6). Foraging strategies during discarding activity differed between species: gulls and Cory's shearwater aggregated and foraged behind the trawlers, whereas the Balearic shearwater was observed further away from the stern of the vessels. The latter showed a lateral foraging behaviour (diving from one side to the other) avoiding the central bird aggregation behind the stern. We observed intra-specific kleptoparasitism for the Cory's shearwater, whereas yellow-legged gulls performed both intra and inter-specific kleptoparasitism (mostly upon diving species which reached sank fish items).

Table 6.6 Number, mean, median, maximum and the percentage of presence (%P) of seabirds attending trawlers are given by season for breeding and non-breeding (noted by *) species.

Species	Winter					Spring					Summer					Autumn				
	N	Mean	Median	Max	% P	N	Mean	Median	Max	% P	N	Mean	Median	Max	% P	N	Mean	Median	Max	% P
Balearic shearwater	49	9.8	2.0	40	100.0	85	4.5	2.0	31	89.5	10	0.63	0.00	3	43.7	1	0.17	0.00	1	16.7
Cory's shearwater	32	6.4	0.0	32	20.0	1349	71.0	40.0	180	100.0	274	17.13	8.00	70	100.0	271	45.17	40.00	110	100.0
European storm-petrel						51	2.7	0.0	20	47.4	8	0.50	0.00	2	37.5					
Mediterranean shag	2	0.4	0.0	1	40.0	5	0.3	0.0	3	10.5	20	1.25	0.00	12	43.7	2	0.33	0.00	1	33.3
Audouin's gull	3	0.6	0.0	3	20.0	254	13.4	5.0	75	100.0	50	3.13	1.50	9	68.7					
Yellow-legged gull	1221	244.2	250.0	450	100.0	3527	185.6	170.0	500	100.0	2539	158.69	140.00	450	100.0	1010	168.33	135.00	300	100.0
Northern gannet*	5	1.0	0.0	4	40.0										2	0.33	0.00	2	16.7	
Great skua*	6	1.2	0.0	3	40.0	8	0.4	0.0	2	36.8										
Mediterranean gull*	1	0.2	0.0	1	20.0															
Lesser black-backed gull*						1	0.1	0.0	1	5.3										
Black tern*						34	1.8	0.0	22	26.3										
Total	1320	264.0	290.0	453		5443	286.5	248.0	530		3060	191.3	161.0	466		1286	214.3	170.0	411	

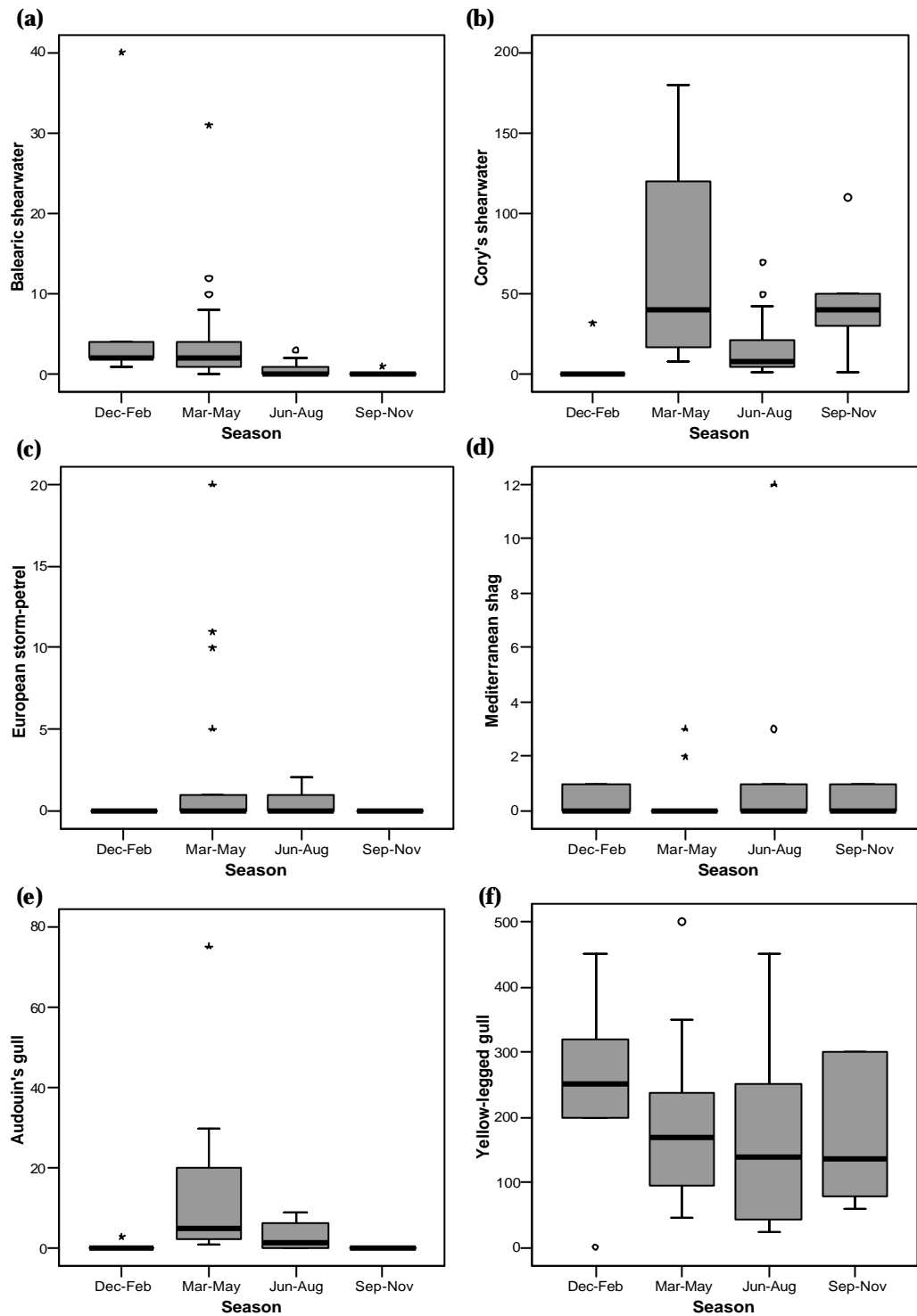


Figure 6.3 Number of breeding species (median, 25%-75% interquartile range, non-outlier range, and outliers) attending trawlers in the study area. Data are presented by season (winter $n = 5$, spring $n = 19$, summer $n = 16$ and autumn $n = 6$) for (a) Balearic shearwaters, (b) Cory's shearwaters, (c) European storm-petrels, (d) Mediterranean shags, (e) Audouin's gulls and (f) yellow-legged gulls.

We found significant seasonal differences (and not a fishing stratum effect) for three of the six breeding species: Balearic shearwater (Kruskal-Wallis test, $H_{3,46} = 19.5$, $p < 0.001$), Cory's shearwater (Kruskal-Wallis test, $H_{3,46} = 17.6$, $p = 0.001$), and Audouin's gull (Kruskal-Wallis test, $H_{3,46} = 21.5$, $p < 0.001$). Balearic shearwaters occurred mainly during winter (pre-breeding stage, maximum 40 birds) and spring (incubation, maximum 31 birds) (Table 6.6; Figure 6.3a). Cory's shearwaters were significantly more abundant during spring (i.e. pre-breeding) and autumn (i.e. end of the breeding season) (maximum of 180 and 110, respectively) (Table 6.6; Figure 6.3b). Audouin's gulls occurred mainly during spring (i.e. laying and incubation periods) (maximum of 75) while they were absent in autumn (Table 6.6; Figure 6.3e). The European storm-petrel and the Mediterranean shag attended trawlers on a very irregular basis, most often being absent or present in small numbers (Figure 6.3c and 6.3d). The yellow-legged gull presented similar (the highest) abundances throughout the year (in all seasons observing maximums over 300 birds) (Table 6.6; Figure 6.3f). In relation to ages, immatures of Mediterranean shags and adults of both Audouin's and yellow-legged gulls were mainly involved in the exploitation of discards (60%, 91% and 67%, respectively).

The main spatial aggregations of Balearic shearwater were found close to Palma, probably attracted by the confluence of trawlers returning to the harbour (Figure 6.4a), whereas larger concentrations of Audouin's gull were recorded in the west coast of Formentera (Figure 6.4e). Cory's shearwaters and yellow-legged gulls were the most abundant species in all areas (Figure 6.4b and 6.4f). Although scarce, the European storm-petrel and the Mediterranean shag were mostly present in south-western Mallorca (Table 6.6; Figure 6.4c, and 6.4d).

During spring, we found overall differences between the observed and expected frequencies of breeding seabirds attending trawlers in south-west Mallorca ($\chi^2_5 = 579.38$, $P < 0.001$, number of hauls = 10). Contrast analyses showed that the Balearic shearwater, the European storm-petrel, the Mediterranean shag and the Audouin's gull were significantly less abundant than expected from their respective population densities. On the contrary, the Cory's shearwater was significantly more abundant in relation to the breeding population in the area, whereas no significant difference was detected for the yellow-legged gull.

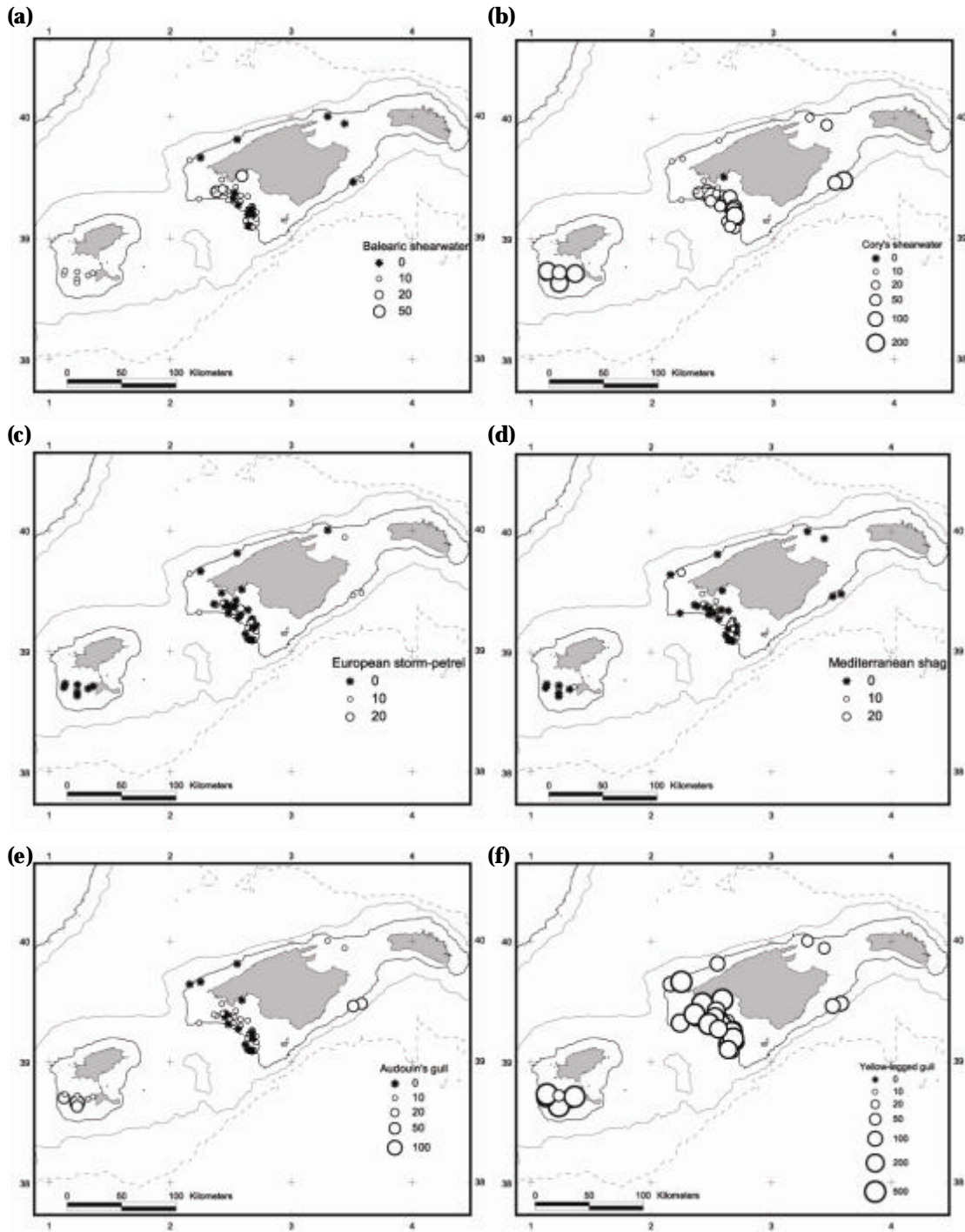


Figure 6.4 Distribution of (a) Balearic shearwaters, (b) Cory's shearwaters, (c) European storm-petrels, (d) Mediterranean shags, (e) Audouin's gulls and (f) yellow-legged gulls per discarding process during the study period. Circles are proportional to the number of birds.

We observed five non-breeding species, previously unreported in association with trawlers at the archipelago (Oro & Ruiz 1997): the Northern gannet, Great skua, Mediterranean gull *Larus melanocephalus*, lesser black-backed gull *L. fuscus* and black tern *Chlidonias niger* (see Table 6.6). We also recorded bottlenose dolphins (*Tursiops truncatus*) in 59% of a total of 29 fishing operations, associated mostly with coastal trawlers. .

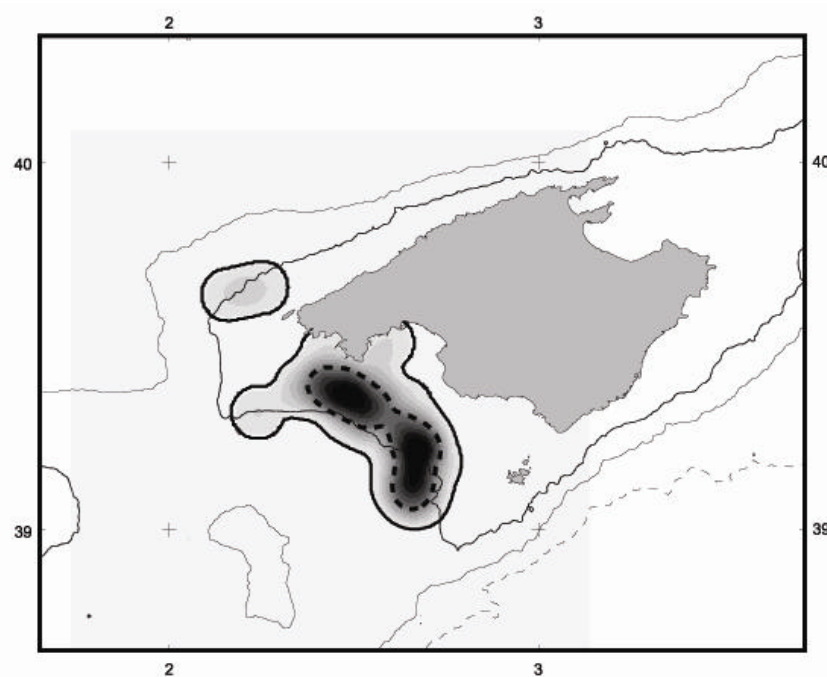


Figure 6.5 Fixed kernel showing the main trawling areas off SW Mallorca, throughout the year. Contours of 50% and 95% probabilities are shown (the core and range lines).

Area of discarding activity

Discarding activity took place mainly in the south-west coast of Mallorca, influencing 12% of the total breeding seabird numbers (in 1991). The core area was located mainly in the continental shelf whereas the range area also comprised coastal waters and the continental slope (see Figure 6.5).

6.5 Discussion

This study provides, for the first time, a year-round picture of seabird attendance to trawlers in the Balearic archipelago, involving both breeding and non-breeding species. Discarding activity was mainly centred in the continental shelf off the south-west coast

of Mallorca, which holds 12% of the total seabird breeding numbers and the main trawling harbour of the archipelago. The highest seabird numbers were recorded, firstly, in both the coastal and shelf fishing strata during winter (and less in summer) and, secondly, when a higher number of powerful trawlers occurred simultaneously. When comparing our seabird abundances attending trawlers at the Balearic archipelago to previous years (Oro & Ruiz 1997), results suggest that discards represented a small proportion of the energetic requirements of the seabird community at the archipelago. Therefore, seabirds do not rely greatly on discards.

By species, the most common breeding species (yellow-legged gulls and Cory's shearwaters) were also the most common following trawlers. Abundances of the yellow-legged gull were proportional to their breeding densities, although it was less abundant than in the Iberian Peninsula (Abelló et al. 2003). For the Cory's shearwater, higher numbers than expected were recorded, and this species was especially abundant during both the pre-breeding stage and the end of the breeding season (see also Arcos 2001, Martínez-Abraín et al. 2002). On the contrary, the remaining breeding species occurred in lower numbers than expected, which suggest that discards were of relative little importance to them. In the case of the endemic Balearic shearwater, studies off the Ebro Delta recorded larger numbers during the breeding season (Arcos & Oro 2002a), possibly related to differences in the number of trawlers operating and the amounts and composition of discards at the Ebro Delta (Oro & Ruiz 1997, Arcos 2001). European storm petrels and Mediterranean shags do not exploit extensively trawling discards (see also Garthe et al. 1996, Oro & Ruiz 1997, Martínez-Abraín et al. 2002), although the occurrence of the latter (both adults and immature birds) was higher than previously reported in the study area (Oro & Ruiz 1997), which suggests that the species has recently incorporated this feeding technique. Although shags are considered strict coastal feeders (Wanless & Harris 1997), birds attended trawlers at distances up to 20 km offshore. Audouin's gulls attended trawlers in lower numbers, relative to the Ebro Delta area, probably as a result of an outcompetition by the yellow-legged gull (the ratio yellow-legged/Audouin's gulls is much higher at the Balearic archipelago, see Arcos et al. 2001, Oro et al. 2006).

In relation to the trawling fishery, an approximate average of 1050 tons was discarded annually in Mallorca during the study period. The fishing stratum influenced the amount of discards, landings and discard ratio, probably related to the geographic situation, geomorphologic characteristics, and bathymetry, being the slope

stratum the most selective (Furness et al. 1988, Carbonell et al. 1998). As previously found in other areas of the western Mediterranean, a strong association appeared between discards and landings, and overall discards corresponded to 74% of landings (range: 9 - 600%) (Oro & Ruiz 1997, Arcos 2001, but see Martínez-Abraín et al. 2002). Discard composition and landings were characterised by a major proportion of fish followed by crustaceans and molluscs (ca. 80%, 15%, and 5%, respectively) (c.f. Moranta et al. 2000).

An ecosystem-based management approach for the western Mediterranean trawling fishery: taking into account seabird conservation

Ecosystem-based fishery management prioritises the whole ecosystem rather than target species, allowing for extraction of marine living resources at sustainable levels for both the stocks and the whole ecosystem (Pikitch 2004). Fisheries may affect the marine environment in numerous ways, and it is particularly important to reduce and avoid its disturbance and degradation, as well as the promotion of irreversible ecological changes (Pikitch et al. 2004, Norse et al. 2005). Fisheries could induce the loss of a species and, in turn, the disappearance of its ecological role may induce the disappearance of many other species through trophic cascade effects, being especially important the function of marine top predators (Coleman & Williams 2002, Soulé et al. 2003, Norse et al. 2005, Sergio et al. 2005).

The western Mediterranean Sea is highly exploited and current levels of both fishing activities and environmental forcing are probably leading important changes in the whole ecosystem (Bianchi & Morri 2000, Tudela 2004). Recently, an integrative ecological model showed that the ecosystem was highly constrained by predators (both natural and anthropogenic) and identified demersal trawling as the most impacting and less effective fishing gear on both target and non-target groups (Coll et al. 2006). Mediterranean fisheries have been managed so far without paying much attention to ecosystem components other than the targeted species (Tudela 2004) and regulations, mainly directed to trawling, are based on effort control (limiting time at sea), minimum landing and mesh sizes. In some cases (e.g. the Mediterranean Iberian Peninsula), time-area closures are implemented, although spatial closures are not always respected and biological criteria for implementing temporal closures are not very clear (see more details in Leonart & Maynou 2003). Future fisheries policies should include measures

such as a comprehensive monitoring and enforcement, by-catch controls, gear restrictions, effective temporal and spatial distribution of fisheries, and marine protected areas.

Fishing activities are of high concern for Mediterranean seabird conservation, even when direct mortality is not considered (Mínguez et al. 2003). For instance, current seabird food provisioning is influenced by fisheries, providing food supplies (e.g. discards), but limiting natural prey availability (i.e. small pelagic fish) and in turn affecting their life-history traits (Oro 1999, Mínguez et al. 2003, Arcos et al. 2006). From seabird's point of view, future fisheries policies (such as the reduction of discards) require, firstly, a precautionary management approach considering the ecological linkage of target species with the seabird community structure and the habitat (Pikitch et al. 2004, Norse et al. 2005). In a scenario without fishing discards, those species making extensive use on this trophic resource could be seriously affected, through potential reduction of survival and breeding success, which would affect in turn seabird community structure (e.g. Oro et al. 1996, Furness & Tasker 2000, Arcos et al. 2001). In the northwestern Mediterranean, there are evidences of the influence of trawling moratorium on seabird breeding performance (see revision in Oro 1999). Nevertheless, we believe that the long-term effects of discard reduction, as a tool for ecosystem restoration, would be beneficial for seabird populations, although predicting the response of seabird communities is complex and it requires long-term time-series to disentangle the confounding effects of other environmental perturbations (Arcos et al. 2004, Frederiksen et al. 2004, Votier et al. 2004). Also, fisheries management should avoid the localised depletion of small pelagic fish (the main prey for most Mediterranean seabirds) in important foraging areas, some of them likely located close to breeding colonies (e.g. Oro 1999, Arcos et al. 2006). Predation by larger and opportunistic seabirds upon more vulnerable sympatric species could be especially dramatic whether discard reduction overlapped with natural prey shortage (c.f. Votier et al. 2004, Arcos et al. 2006). Moreover, seabird attendance may differ locally in relation to breeding densities, oceanographic habitat, and local trawling fishery characteristics such as is the case of the western Mediterranean seabird community. All those factors make complex to predict seabird response to fishing regulation in the Mediterranean. Additionally, it is important to bear in mind that seabirds are structured in metapopulations, thus local actions would need to consider the global consequences (Mínguez et al. 2003, Oro 2003). Finally, hot-spot areas of

diversity such as oceanographic fronts should be of high conservation concern deserving special protection since they aggregate predators and prey (Norse et al. 2005). Fisheries management should pay particular attention to endangered and protected species, including their ecological role in the ecosystem functioning (Pikitch et al. 2004). Therefore, both hot-spot diversity areas and endangered species protection should also be conservation priorities in ecosystem-based management (Meffe & Carroll 1994).

The critically endangered Balearic shearwater, the most threatened Mediterranean seabird, is a good example of the urgent need of an ecosystem-based management of Mediterranean fisheries (Arcos et al. 2006). The unsustainable high adult mortality is presumably due to entanglement on fishing gears (Oro et al. 2004b), while the foraging ecology of the species is closely related to both pelagic prey distribution and trawling discards availability, which affect its breeding performance and distribution in the western Mediterranean (Louzao et al. 2006a, 2006b). Moreover, it is associated to hot-spot areas of diversity such as oceanographic fronts in the Ebro Delta area (western Mediterranean), where a potential marine protected area was identified on the basis of the dynamic nature of these fronts (Louzao et al. 2006b). However, the establishment of marine protected areas is rather controversial and it could generate socio-political pressure since these areas hold important fishing grounds (Demestre et al. 1998).

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References

- Abelló P, Arcos JM, Gil de Sola L (2003) Geographical patterns of seabird attendance to a research trawler along the Iberian Mediterranean coast. *Scientia Marina* 67S2:69-75
- Aguilar JS (1991) Resum de l'atlas d'ocells marins de les Balears, 1991. *Anuari Ornitológic de les Balears* 6:17-28
- Alverson DL (1998) Discarding practices and unobserved fishing mortality in marine fisheries: an update. Washington Sea Grant Programme. University of Washington, 82 pp
- Anderson DJ (1982) The home range: a new non-parametric estimation technique. *Ecology* 63:103-112
- Arcos JM (2001) Foraging ecology of seabirds at sea: significance of commercial fisheries in the NW Mediterranean. Doctoral dissertation. Universitat de Barcelona. Available: http://tdcat.cesca.es/TESIS_UB/AVAILABLE/TDX-0219102-114337/TOL38.pdf (March 2002)
- Arcos JM, Oro D (2002a) Significance of fisheries discards for a threatened Mediterranean seabird, the Balearic shearwater *Puffinus mauretanicus*. *Marine Ecology Progress Series* 239:209-220
- Arcos JM, Oro D (2002b) Significance of nocturnal purse seine fisheries for seabirds: a case study off the Ebro Delta (NW Mediterranean). *Marine Biology* 141:277-286
- Arcos JM, Sol D, Oro D (2001) Competition between the yellow-legged gull *Larus cachinnans* and Audouin's gull *Larus audouinii* associated with commercial fishing vessels: the influence of season and fishing fleet. *Marine Biology* 139:807-816
- Arcos JM, Louzao M, Oro D (2004) Management of Mediterranean fisheries and impact on the marine ecosystem: how to account for top predators? *Rapp. Comm. Int. Mer Médit.* 37:307
- Arcos JM, Louzao M, Oro D (2006) Fishery Ecosystem Impacts and Management in the Mediterranean: Seabirds Point of View. In: Nielsen J, Dodson J, Friedland K,

- Hamon T, Hughes N, Musick J, Verspoor E (Eds) Proceedings of the Fourth World Fisheries Congress: Reconciling Fisheries with Conservation. American Fisheries Society, Symposium 49, Bethesda, Maryland, *in press*.
- Bianchi CN, Morri C (2000) Marine biodiversity of the Mediterranean Sea: situation, problems and prospects for future research. *Marine Pollution Bulletin* 40:367-376
- Brothers N, Gales R, Reid T (1999) The influence of environmental variables and mitigation measures on seabird catch rates in the Japanese tuna longline fishery within the Australian Fishing Zone, 1991–1995. *Biological Conservation* 88:85-101
- Camphuysen CJ, Calvo B, Durinck J, Ensor K, Follestad A, Furness RW, Garthe S, Leaper G, Skov H, Tasker ML, Winter CJN (1995) Consumption of discards by seabirds in the North Sea. Final report EC DG XIV. Netherlands Institute for Sea Research, Texel. 202 pp.+LVI
- Carbonell A, Martin P, De Ranieri S, WEDIS team (1998) Discards of the western Mediterranean trawl fleets. *Rapp. Comm. Int. Mer Médit.* 35:392-393
- Carbonell A, Alemany F, Merella P, Quetglas A, Román E (2003) The by-catch of sharks in the western Mediterranean (Balearic Islands) trawl fishery. *Fisheries Research* 61:7-18
- Coleman FC, Williams SL (2002) Overexploiting marine ecosystem engineers: potential consequences for biodiversity. *Trends in Ecology and Evolution* 17:40-44
- Coll M, Palomera I, Tudela S, Sardà F (2006) Trophic flows, ecosystem structure and fishing impacts in the South Catalan Sea, Northwestern Mediterranean. *Journal of Marine Systems* 59:63-96
- COM (2002) 656. Commission of the European Communities. Communication from the Commission to the Council and the European Parliament on the Community Action Plan to reduce discards of fish, Brussels, Belgium
- Demestre M, Lleonart J, Martin P, Recasens L, Sánchez P (1988) La pesca en Cataluña. *FAO Rapport sur les Pêches* 395:101-103
- Farrugio H, Oliver P, Biagi F (1993) An overview of the history, knowledge, recent and future trends in Mediterranean fisheries. *Scientia Marina* 57 (2-3):105-119
- Furness RW (2003) Impacts of fisheries on seabird communities. *Scientia Marina* 67S2:33-45

- Furness RW, Ainley (1984) Threats to seabird populations presented by commercial fisheries. ICBP Technical Publication 2:701-708
- Furness RW, Camphuysen CJ (1997) Seabirds as monitors of the marine environment. ICES Journal of Marine Science 54:726-737
- Furness RW, Tasker ML (2000) Seabird-fishery interactions: quantifying the sensitivity of seabirds to reduction of key areas for sensitive seabirds in the North Sea. Marine Ecology Progress Series 202:253-264
- Furness RW, Hudson AV, Ensor K (1988) Interactions between scavenging seabirds and commercial fisheries around the British Isles. In: Burger J (Ed) Seabirds & Other Marine Vertebrates. Competition, Predation & Other Interactions. Columbia University Press, USA, pp.240-268
- Garthe S (1997) Influence of hydrography, fishing activity, and colony location on summer seabird distribution in the south-eastern North Sea. ICES Journal of Marine Science 54:566-577
- Garthe S, Camphuysen CJ, Furness RW (1996) Amounts of discards by commercial fisheries and their significance as food for seabirds in the North Sea. Marine Ecology Progress Series 136:1-11
- Grosbois V, Thompson PM (2005) North Atlantic climate variation influences survival in adult fulmars. Oikos 109:273-290
- Guijarro B, Massutí E (2006) Selectivity of diamond- and square-mesh codends in the deepwater crustacean trawl fishery off the Balearic Islands (western Mediterranean). ICES Journal of Marine Science 63:52-67
- Heppell SS, Heppell SA, Read AJ, Crowder LB (2005) Effects of fishing long-lived marine organisms. In: Norse EA, Crowder LB (Eds) Marine Conservation Biology. Island Press, Washington, USA, pp. 211-231
- Hooge PN, Eichenlaub B (1997) Animal movement extension to ArcView, version 2.04. Alaska Biological Science Center, US Geological Survey, Anchorage, AK
- Huppopp O, Wurm S (2000) Effects of winter fishery activities on resting numbers, food and body condition of large gulls *Larus argentatus* and *L. marinus* in the south-eastern North Sea. Marine Ecology Progress Series 194:241-247
- Kahru M (2000) Windows Image Manager - Image display and analysis program for Microsoft Windows with special features for satellite images. Available at: <http://wimsoft.com>

- Krebs CJ (2001) *Ecology: the experimental analysis of distribution and abundance*. Benjamin Cummings, San Francisco, CA
- Lewison RL, Crowder LB, Read AJ, Freeman SA (2004) Understanding impacts of fisheries bycatch on marine megafauna. *Trends in Ecology and Evolution* 19:598-604
- Lleonart J, Maynou F (2003) Fish stock assessment in the Mediterranean: state of the art. *Scientia Marina* 67S1:37-49
- Louzao M, Hyrenbach KD, Arcos JM, Abelló P, Gil de Sola L, Oro D (2006a) Oceanographic habitat of an endangered Mediterranean Procellariiform: implications for marine protected areas. *Ecological Applications* (in press)
- Louzao M, Igual JM, McMinn M, Aguilar JS, Triay R, Oro D (2006b) Small pelagic fish, trawling discards and breeding performance of the critically endangered Balearic Shearwater: improving conservation diagnosis. *Marine Ecology Progress Series* 318:247-254
- Martínez-Abraín A, Oro D, Maestre R (2002) Demersal trawling waste as a food source for western Mediterranean seabirds during the summer. *ICES Journal of Marine Science* 59:529-537
- Massutí E, Reñones O (2005) Demersal resource assemblages in the trawl fishing stratum off the Balearic Islands (western Mediterranean). *Scientia Marina* 69:167-181
- Meffe GK, Carroll RC (1997) *Principles of Conservation Biology*, 2nd ed. Sinauer Associates, Sunderland, MA
- Mínguez E, Oro D, De Juana E, Martínez-Abraín A (2003) Mediterranean seabird conservation: what can we do? *Scientia Marina* 67S2:3-6
- Moranta J, Massutí E, Morales-Nin B (2000) Fish catch composition of the deep-sea decapod crustacean fisheries in the Balearic Islands (western Mediterranean). *Fisheries Research* 45:253-264
- Myers RA, Worm B (2003) Rapid worldwide depletion of predatory fish communities. *Nature* 423:280-283
- Myers RA, Worm B (2005) Extinction, survival or recovery of large predatory fishes. *Transactions of the Royal Society of London, Series B* 360:13-20
- NGDC (National Geophysical Data Center) (1998) Data Announcement 88-MGG-02, Digital relief of the surface of the Earth. Boulder: National Geophysical Data Center (www.ngdc.noaa.gov/mgg/global/seltopo.html)

- Norse EA, Crowder LB, Gjerde K, Hyrenbach D, Roberts CM, Safina C, Soulé ME (2005) Place-based ecosystem management in the open ocean. In: Norse,EA, Crowder LB (Eds) Marine conservation biology. Island Press, Washington, USA, pp. 302-327
- Oro D (1996) Interspecific kleptoparasitism in Audouin's Gull *Larus audouinii* at the Ebro Delta, northeast Spain: a behavioural response to low food availability. *Ibis* 138:218-221
- Oro D (1999) Trawler discards: a threat or a resource for opportunistic seabirds? In: Adams NJ, Slotow RH (Eds) Proceedings of the 22nd International Ornithological Congress, Durban. Birdlife South Africa, Johannesburg, pp. 717-730
- Oro D, Ruiz X (1997) Exploitation of trawler discards by breeding seabirds in the north-western Mediterranean: differences between the Ebro Delta and the Balearic Islands areas. *ICES Journal of Marine Science* 54:695-707
- Oro D, Furness RW (2002) Influence of food availability and predation on survival of kittiwakes. *Ecology* 83:2516-2528
- Oro D, Bosch M, Ruiz X (1995) Effects of a trawling moratorium on the breeding success of the yellow-legged gull *Larus cachinnans*. *Ibis* 137:347-349
- Oro D, Jover L, Ruiz X (1996) Influence of trawling activity on the breeding ecology of a threatened seabird, Audouin's gull *Larus audouinii*. *Marine Ecology Progress Series* 139:19-29
- Oro D, Cam E, Pradel R, Martínez-Abraín A (2004a) Influence of food availability on demography and local population dynamics in a long-lived seabird. *Proceedings of the Royal Society of London Series B* 271:387-396
- Oro D, Aguilar JS, Igual JM, Louzao M (2004b) Modelling demography and extinction risk in the endangered Balearic shearwater. *Biological Conservation* 116:93-102
- Oro D, Martínez-Abraín A, Paracuellos M, Nevado JC, Genovart M (2006) Influence of density dependence on predator-prey seabird interactions at large spatio-temporal scales. *Proceedings of the Royal Society of London Series B* 273:379-383
- Pauly D, Christensen V, Guenette S, Pitcher TJ, Sumaila UR, Walters CJ, Watson R, Zeller D (2002) Towards sustainability in world fisheries. *Nature* 418:689-695
- Pikitch EK, Santora C, Babcock EA, Bakun A, Bonfil R, Conover DO, Dayton P, Doukakis P, Fluharty D, Heneman B, Houde ED, Link J, Livingston PA, Mangel

- M, McAllister MK, Pope J, Sainsbury KJ (2004) Ecosystem-based fishery management. *Science* 305:346-347
- Pinot JM, López-Jurado JL, Riera M (2002) The CANALES experiment (1996-1998). Interannual, seasonal, and mesoscale variability of the circulation in the Balearic Channels. *Progress in Oceanography* 55:335-370
- Rebassa M (2003) Observacions d'aus marines des de punts estratègics de la costa mallorquina. *Anuari Ornitològic de les Balears* 18:39-49
- Reeves SA, Furness RW (2002) Net loss-seabirds gain? Implications of fisheries management for seabirds scavenging discards in the northern North Sea. RSPB, Sandy
- SAS Institute (2000) SAS/STAT® software: User's guide. Cary, North Carolina, USA
- Schneider DC (1997) Habitat selection by marine birds in relation to water depth. *Ibis* 139:175-178
- Selkirk SW, Bishop ID (2002) Improving and extending home range and habitat analysis by integration with a Geographic Information System. *Transactions in GIS* 6(2):151-159
- Sergio F, Newton I, Marchesi L (2005) Top predators and biodiversity. *Nature* 436:192
- Stenhouse I, Montevecchi WA (1999) Indirect effects of the availability of forage fish and fisheries discards: gull predation on breeding storm-petrels. *Marine Ecology Progress Series* 184:303-307
- Tasker ML, Camphuysen CJ, Cooper J, Garthe S, Montevecchi WA, Blaber SJM (2000) The impacts of fishing on marine birds. *ICES Journal of Marine Science* 57:531-547
- Tudela S (2004) Ecosystem effects of fishing in the Mediterranean: an analysis of the major threats of fishing gear and practices to biodiversity and marine habitats. General Fisheries Commission for the Mediterranean (FAO). *Studies and Reviews* 74
- Viada C (2006) *Llibre vermell dels Vertebrats de les Balears* (3^a Ed). Document inèdit. Secció d'Espècies Protegides, Conselleria de Medi Ambient del Govern de les Illes Balears
- Votier SC, Furness RW, Bearhop S, Crane JE, Caldow RWG, Catry P, Ensor K, Hamer KC, Hudson AV, Kalmbach E, Klomp NI, Pfeiffer S, Phillips RA, Prieto I, Thompson DR (2004) Changes in fisheries discard rates and seabird communities. *Nature* 427:727-730

- Wanless S, Harris MP (1997) Shag *Phalacrocorax aristotelis*. BWP Update 1:3-13
- Worton BJ (1989) Kernel methods for estimating the utilization distribution in home-range studies. *Ecology* 70:164-168
- Yen PPW, Sydeman WJ, Hyrenbach KD (2004) Marine bird and cetacean associations with bathymetric habitats and shallow-water topographies: implications for trophic transfer and conservation *Journal of Marine Systems* 50:79-99
- Zar JH (1999) *Biostatistical Analysis*. Prentice Hall, New Jersey
- Zotier R, Bretagnolle V, Thibault JC (2000) Biogeography of the marine birds of a confined sea, the Mediterranean. *Journal of Biogeography* 26:297-313

Appendix 1. Discard composition from the trawling fishery of the Balearic Islands. Number and mean (\pm SD) of 66 taxonomic groups are given by fishing stratum (n = 19 fishing operations).

Family	Common name	Coastal		Shelf		Slope		Total	
		N	Length (\pm SD)	N	Length (\pm SD)	N	Length (\pm SD)	N	Length (\pm SD)
Scyliorhinidae	Blackmouth catshark (<i>Galeus melastomus</i>)			265	222.9 (\pm 35.6)	167	238.4 (\pm 85.1)	432	228.9 (\pm 60.2)
Scyliorhinidae	Small-spotted catshark (<i>Scyliorhinus canicula</i>)	19	286.6 (\pm 32.2)	90	218.4 (\pm 42.0)	1	319	110	231.1 (\pm 48.5)
Dalatiidae	Velvet belly lantern shark (<i>Etmopterus spinax</i>)			7	163.1 (\pm 12.4)	21	198.7 (\pm 49.6)	28	189.8 (\pm 45.9)
Rajidae	Brown ray (<i>Raja miraletus</i>)	6	368.0 (\pm 29.3)					6	368.0 (\pm 29.3)
Rajidae	Rough ray (<i>Raja radula</i>)	1	326					1	326
Rajidae	Thornback ray (<i>Raja clavata</i>)	1	350	1	268			2	309.0 (\pm 226.7)
Chimaeridae	Rabbit fish (<i>Chimaera monstrosa</i> *)			1	93	1	90	2	91.5 (\pm 2.1)
Clupeidae	Gilt sardine (<i>Sardinella aurita</i>)	15	202.7 (\pm 13.2)					15	202.7 (\pm 13.2)
Clupeidae	Pilchard (<i>Sardina pilchardus</i>)	26	155.4 (\pm 37.8)	4	184.0 (\pm 7.0)			30	159.2 (\pm 36.5)
Congridae	Conger eel (<i>Conger coger</i>)					1	397	1	397
Nettastomatidae	Blackfin sorcerer (<i>Nettastoma melanurum</i>)					1	460	1	460
Ophichthidae	Serpent eel (<i>Ophisurus serpens</i>)					1	464	1	464
Nemichthyidae	Slender snipe eel (<i>Nemichthys scolopaceus</i>)					1	907	1	907
Notacanthidae	Shortfin spiny eel (<i>Notacanthus bonapartei</i>)					10	210.2 (\pm 38.4)	10	210.2 (\pm 38.4)
Notacanthidae	Smallmouth spiny eel (<i>Polyacanthonotus rissoanus</i>)					1	225	1	225
Argentinidae	Small-toothed argentine (<i>Glossanodon leioglossus</i>)			8	133.5 (\pm 14.4)			8	133.5 (\pm 14.5)
Stomiidae	Sloane's viperfish (<i>Chauliodus sloani</i>)					2	233.0 (\pm 2.8)	2	233.0 (\pm 2.8)
Stomiidae	Scaly dragonfish (<i>Stomias boa</i>)			2	191.0 (\pm 12.7)	1	192	3	191.3 (\pm 9.0)
Myctophidae	Glacier lanternfish (<i>Benthoosema glaciale</i>)					1	130	1	130
Myctophidae	Jewel lanternfish (<i>Lampanyctus crocodilus</i>)			8	82.2 (\pm 13.5)	184	117.6 (\pm 31.6)	192	116.2 (\pm 31.8)
Clorophthalmidae	Shortnose greeneye (<i>Chlorophthalmus agassizi</i>)			38	113.0 (\pm 17.5)	6	116.0 (\pm 7.8)	44	113.4 (\pm 16.4)
Lophiidae	Angler (<i>Lophius sp.</i>)	2	100.0 (\pm 0.0)	1	210			3	136.7 (\pm 63.5)

Section II: Seabird attendance to trawling at the Balearic Islands

Family	Common name	Coastal		Shelf		Slope		Total	
		N	Length (\pm SD)	N	Length (\pm SD)	N	Length (\pm SD)	N	Length (\pm SD)
Merlucciidae	Hake (<i>Merluccius merluccius</i>)	2	170.0 (\pm 0.0)	3	129.7 (\pm 17.6)			5	145.8 (\pm 76.1)
Gadidae	Silvery pout (<i>Gadiculus argenteus</i>)			55	91.0 (\pm 12.3)	3	91.3 (\pm 4.7)	58	91.0 (\pm 308.9)
Gadidae	Blue whiting (<i>Micromesistius poutassou</i>)					1	210	1	210
Gadidae	Blue ling (<i>Molva dypterygia</i>)					1	480	1	480
Gadidae	Greater forkbeard (<i>Phycis blennoides</i>)			19	158.8 (\pm 9.3)	38	145.6 (\pm 34.0)	57	150.0 (\pm 28.8)
Gadidae	Mediterranean bigeye rockling (<i>Gaidropsarus biscayensis</i>)					1	99	1	99
Gadidae	Common mora (<i>Mora moro</i>)					1	140	1	140
Macrouridae	Glasshead grenadier (<i>Hymenocephalus italicus</i>)			2	46.5 (\pm 4.9)	27	36.6 (\pm 4.3)	29	37.3 (\pm 4.9)
Macrouridae	Hollowsnout grenadier (<i>Caelorinchus caelorhincus</i>)			20	45.4 (\pm 20.5)	47	59.1 (\pm 64.5)	67	55.0 (\pm 55.3)
Macrouridae	Common Atlantic grenadier (<i>Nezumia aequalis</i>)			2	45.0 (\pm 4.2)	21	41.6 (\pm 26.5)	23	41.9 (\pm 25.3)
Carapidae	Pearl fish (<i>Carapus acus</i>)	3	173.3 (\pm 4.2)					3	173.3 (\pm 4.2)
Trachichthyidae	Mediterranean slimehead (<i>Hoplostethus mediterraneus</i>)			3	147.7 (\pm 7.8)	21	151.1 (\pm 44.3)	24	150.7 (\pm 41.4)
Zeidae	John dory (<i>Zeus faber</i>)	1	117	1	74			2	95.5 (\pm 30.4)
Caproidae	Boarfish (<i>Capros aper</i>)	27	73.5 (\pm 15.8)	30	72.2 (\pm 13.0)	3	88.3 (\pm 7.6)	60	73.6 (\pm 14.4)
Macroramphosidae	Longspine snipefish (<i>Macroramphosus scolopax</i>)	60	123.4 (\pm 10.7)	30	111.7 (\pm 13.3)	6	142.7 (\pm 29.7)	96	120.9 (\pm 15.2)
Sebastidae	Blackbelly rosefish (<i>Helicolenus dactylopterus</i>)			44	106.4 (\pm 23.5)	12	103.8 (\pm 9.5)	56	105.8 (\pm 21.2)
Triglidae	Large-scaled gurnard (<i>Lepidotrigla cavillone</i>)	80	110.8 (\pm 13.8)	17	125.2 (\pm 19.2)			97	113.3 (\pm 15.8)
Triglidae	Streaked gurnard (<i>Triglopterus lastoviza</i>)	1	120					1	120
Triglidae	East Atlantic red gurnard (<i>Aspitrigla cuculus</i>)	6	153.5 (\pm 11.5)	17	115.8 (\pm 19.0)			23	125.6 (\pm 24.1)
Peristediidae	African armoured searobin (<i>Peristedion cataphractum</i>)	2	218.5 (\pm 12.0)					2	218.5 (\pm 12.0)
Serranidae	Brown comber (<i>Serranus hepatus</i>)	16	92.7 (\pm 5.7)					16	92.7 (\pm 5.7)
Carangidae	Mediterranean horse mackerel (<i>Trachurus mediterraneus</i>)	367	165.5 (\pm 10.9)	2	174.5 (\pm 9.2)			369	165.6 (\pm 10.9)

Section II: Seabird attendance to trawling at the Balearic Islands

Family	Common name	Coastal		Shelf		Slope		Total	
		N	Length (\pm SD)	N	Length (\pm SD)	N	Length (\pm SD)	N	Length (\pm SD)
Carangidae	Atlantic horse mackerel (<i>Trachurus trachurus</i>)	24	176.13 (\pm 21.20)	6	217.83 (\pm 13.11)			30	184.47 (\pm 25.96)
Mullidae	Striped red Mollet (<i>Mullus surmuletus</i>)	2	128.0 (\pm 9.9)	1	123			3	126.3 (\pm 7.6)
Sparidae	Bogue (<i>Boops boops</i>)	296	173.1 (\pm 25.0)	34	204.2 (\pm 12.4)			330	176.3 (\pm 25.8)
Sparidae	Common Pandora (<i>Pagellus erythrinus</i>)	15	117.4 (\pm 15.0)					15	117.4 (\pm 71.6)
Sparidae	Blackspot seabream (<i>Pagellus bogaraveo</i>)	1	185					1	185
Sparidae	Axillary seabream (<i>Pagellus acarne</i>)	14	125.7 (\pm 8.9)					14	125.7 (\pm 8.9)
Centracanthidae	Curled picarel (<i>Centracanthus cirrus</i>)	72	180.4 (\pm 15.4)			2	179.5 (\pm 2.1)	74	180.4 (\pm 15.2)
Centracanthidae	Blotched picarel (<i>Spicara maena</i>)	6	168.2 (\pm 15.9)					6	168.2 (\pm 15.9)
Centracanthidae	Picarel (<i>Spicara smaris</i>)	19	141.4 (\pm 10.5)	5	133.0 (\pm 19.7)			24	139.7 (\pm 12.9)
Cepolidae	Red bandfish (<i>Cepola macrophthalma</i>)	1	348					1	348
Blenniidae	Butterfly blenny (<i>Blennius ocellaris</i>)	2	121.0 (\pm 18.4)					2	121.0 (\pm 321.5)
Callionymidae	<i>Synchiropus phaeton</i>	22	175.9 (\pm 24.2)	85	154.1 (\pm 29.7)			107	158.6 (\pm 29.9)
Trichiuridae	Silver scabbardfish (<i>Lepidopus caudatus</i>)			10	477.3 (\pm 70.9)	6	437.5 (\pm 93.6)	16	462.4 (\pm 79.6)
Citharidae	Atlantic spotted flounder (<i>Citharus linguatula</i>)	4	173.5 (\pm 31.7)	9	158.0 (\pm 14.9)			13	162.8 (\pm 21.3)
Scophthalmidae	Fourspotted megrim (<i>Lepidorhombus boscii</i>)			100	130.3 (\pm 16.1)	11	124.4 (\pm 14.5)	111	129.7 (\pm 16.0)
Bothidae	Rüppell's scaldback (<i>Arnoglossus rueppelli</i>)	3	127.3 (\pm 2.5)					3	127.3 (\pm 2.5)
Bothidae	Scaldfish (<i>Arnoglossus laterna</i>)	3	89.7 (\pm 69.1)	11	117.8 (\pm 7.4)			14	111.8 (\pm 30.3)
Bothidae	Imperial scaldfish (<i>Arnoglossus imperiales</i>)	2	124.0 (\pm 65.0)					2	124.0 (\pm 65.0)
Bothidae	Thor's scaldfish (<i>Arnoglossus thori</i>)	5	97.6 (\pm 7.0)					5	97.6 (\pm 7.0)
Soleidae	Thickback sole (<i>Microchirus variegatus</i>)	2	151.0 (\pm 1.4)			3	118.3 (\pm 30.9)	5	131.4 (\pm 219.4)
Soleidae	Whiskered sole (<i>Monochirus hispidus</i>)			1	167	1	127	2	147.0 (\pm 28.3)
Cynoglossidae	Tongue sole (<i>Symphurus nigrescens</i>)					5	70.8 (\pm 35.7)	5	70.8 (\pm 35.7)

Conclusiones

Conclusiones

La conservación de la pardela balear *Puffinus mauretanicus* y su estatus han despertado el interés de la Administración e investigadores en los últimos años. Esta tesis doctoral ha intentado avanzar en el conocimiento de la biología de la especie, determinando algunos de los rasgos de historia de vida más importantes para su conservación y describiendo la asociación al hábitat de la especie en un ecosistema modificado. Así, a continuación se destacan las principales conclusiones de esta tesis doctoral centrada en la Biología de la Conservación de la pardela balear.

Conservación en las colonias

1. Por primera vez, se realizó una diagnosis de conservación que confirmó el declive poblacional de la especie. Un estudio demográfico detallado permitió determinar los parámetros demográficos que afectan negativamente a la tasa de crecimiento poblacional: la mortalidad adulta es la principal causa del declive poblacional, mientras que los valores actuales de éxito reproductor (otro parámetro demográfico de potencial relevancia para la conservación de la pardela balear) y los factores que pueden influir en este parámetro (la disponibilidad de alimento) no son cuestiones preocupantes para la conservación de la especie, al menos en la actualidad y en las colonias de estudio. También se ha visto que la estocasticidad demográfica juega un papel más importante en la tasa de crecimiento poblacional que la estocasticidad ambiental, fenómeno típico en poblaciones pequeñas.

2. El análisis de viabilidad de la población mostró que las probabilidades de extinción de la pardela balear son extremadamente altas. El tiempo medio de extinción estimado clasificaba a la especie en Peligro Crítico, según criterios de IUCN, la categoría de amenaza mayor para una especie no extinta en libertad. Las estrategias de conservación prioritarias deben reducir la mortalidad adulta e identificar los factores causantes de la misma. En las colonias de estudio no se detectaron depredadores introducidos, gracias al control de éstos, por lo que las causas de mortalidad provienen principalmente de las zonas de alimentación (probablemente debido a la interacción con artes de pesca). Sin embargo, existen evidencias de que algunas especies introducidas (gatos salvajes) son una amenaza importante para la supervivencia adulta (en otras colonias no estudiadas en el segundo capítulo). Así, las políticas de conservación deberían centrarse en acciones a largo plazo y programas de seguimiento efectivos para mantener y continuar con la protección del hábitat de nidificación y estimar el impacto real y reducir la mortalidad en artes de pesca, como valorar el impacto de otros factores (derrames de hidrocarburos o la baja disponibilidad de alimento). Las acciones prioritarias se han de ejecutar a corto plazo para asegurar la supervivencia y preservación de la pardela balear y de otras aves marinas endémicas de la región mediterránea, tanto en el Mediterráneo occidental como en el Atlántico norte. Para ello, es primordial la inclusión de la especie en los convenios de conservación internacionales e implicar a las agencias de conservación.

3. No existen estimas poblacionales robustas de pardela balear y esto puede ser un problema a la hora de realizar un diagnóstico fiable en cuanto al tiempo estimado hasta la extinción. Aún así, se asume que en este estudio la fiabilidad de las probabilidades de extinción y tasas de crecimiento es alta ya que el parámetro que más influye en la tasa de crecimiento poblacional, la supervivencia, se estimó con un intervalo de confianza estrecho sugiriendo que las predicciones de los modelos fueron suficientemente exactas. Cuando las densidades poblacionales son difíciles de estimar, el cálculo de parámetros demográficos y su uso en el modelado de las poblaciones son cruciales para valorar las tendencias y elaborar una diagnosis de conservación de especies amenazadas. Incluir más años de seguimiento en el estudio demográfico hubiera proporcionado mayor robustez a la estima de supervivencia e incluso hubiera podido identificar variables ambientales (clima, alimento) e individuales (sexo, edad) que influyen en la misma.

4. La comparación de rasgos de historia de vida entre poblaciones de la misma especie puede darnos una idea de las adaptaciones a las condiciones ecológicas locales. Por ello, realizamos un análisis espacio-temporal de parámetros reproductores (productividad y tamaño del huevo) de diferentes poblaciones locales de pardela balear. La productividad varió con los años y de manera similar en todas las colonias. Así, la disponibilidad de alimento, tanto de pequeños peces pelágicos como de descartes de la pesca de arrastre en la zona del delta del Ebro (principal zona de alimentación de la pardela balear), también influyó positivamente en la productividad. La variabilidad temporal de la productividad probablemente fue debida a los cambios anuales en la disponibilidad de pequeños peces pelágicos, que exhiben una estocasticidad temporal, y descartes pesqueros. El periodo de incubación resultó ser el estadio más crítico para el éxito reproductor y éste pudo verse influido por la condición de la hembra durante la época previa a la puesta debido a una variabilidad temporal en la disponibilidad de alimento.

5. En cuanto al tamaño del huevo, se vio que este rasgo de historia de vida variaba entre poblaciones locales vecinas (incluso a distancias mínimas de 100 m) y no con los años, resultado inesperado a una escala espacial tan pequeña, ya que las colonias de estudio presentaban características ecológicas similares. La variabilidad espacial en el tamaño del huevo podría explicarse parcialmente por diferencias interpoblacionales en el nivel trófico en el cual se alimentan los adultos reproductores (valores promedio del isótopo $\delta^{15}\text{N}$). Tanto las diferencias interpoblacionales en el nivel trófico como en el tamaño del huevo mostraron el mismo patrón general, es decir, los huevos más grandes estarían relacionados con valores promedio de $\delta^{15}\text{N}$ mayores, sugiriendo una conexión nutricional entre ambos. Al mismo tiempo los dos factores, el nivel trófico y el tamaño del huevo podrían estar influenciados por el tamaño corporal parental (potencial indicador de un componente genético) puesto que éste podría mediar en la selección de las presas y condicionar el tamaño del huevo. Por tanto, factores genéticos y ecológicos podrían interactuar para dar lugar a las mencionadas diferencias.

6. Los valores promedio del isótopo $\delta^{13}\text{C}$ sugerían que los adultos reproductores de las diferentes poblaciones locales tenían hábitos alimentarios similares en cuanto al tipo de cadena trófica explotada (bentónica vs. pelágica).

7. La variación temporal de los dos parámetros reproductores analizados (productividad y tamaño del huevo) presentan diferentes patrones, probablemente relacionado con la diferente ventana temporal que afecta a ambos, en relación a la disponibilidad de alimento. Es decir, el tamaño del huevo sería un rasgo evolutivo más conservador que la productividad, ya que la formación del primero representa una inversión energética a más corto plazo comparado con el periodo más largo que afecta al segundo. En cuanto a la relación entre ambos parámetros, se contrastó la hipótesis de que huevos más grandes tenían una mayor probabilidad de tener éxito reproductor. Sin embargo, el tamaño del huevo no afectó significativamente a la productividad sugiriendo que los huevos más grandes pueden ser ventajosos únicamente en condiciones adversas, mientras que la calidad parental podría afectar enormemente la productividad en cualesquiera condiciones.

Conservación en el mar

8. Para la conservación de especies amenazadas es de vital importancia conocer su asociación al hábitat e implementar medidas de conservación como las áreas marinas protegidas. El área de campeo de la pardela balear, durante la época reproductora, se localizó en aguas costeras de la plataforma continental Ibérica caracterizadas por sistemas frontales (zonas de agregación de depredadores y presas) en áreas cercanas a las colonias de cría, entre 38.0° y 42.3°N. Dentro del área de campeo, se identificaron las zonas de alimentación más importantes, caracterizadas por altas densidades de aves: las aguas muy productivas cercanas al delta del Ebro y el entorno del cabo de la Nao. Estos resultados sugieren que los patrones de distribución observados, que también mostraron una variabilidad interanual sustancial, eran respuesta a diversos procesos oceanográficos como plumas fluviales, fenómenos de mesoescala y cañones submarinos que incrementan la productividad en zonas localizadas y agregan las presas.

9. Los modelos predictivos resultaron una herramienta útil, y con un gran potencial, para determinar la asociación al hábitat de la pardela balear tanto en áreas muestreadas como no muestreadas, ya que el hábitat oceanográfico predicho por el modelo final concordó con el área de campeo descrito a partir de los datos observados, e incluso identificó las aguas cercanas a las colonias de cría en las islas Baleares como hábitat

adecuado para la especie. Sin embargo, hay que ser cautelosos con esta aproximación porque asume que la asociación al hábitat definida en un área biogeográfica es extrapolable a otras no muestreadas. Por tanto, es necesario realizar campañas oceanográficas estandarizadas en diferentes condiciones ambientales para validar los modelos de asociación al hábitat dentro del área muestreada, y en las áreas predichas por los modelos predictivos, especialmente interesante en el caso de especies amenazadas. Los muestreos estandarizados también son imprescindibles para identificar tendencias estacionales y anuales (tendencias poblacionales).

10. Aunque se conocía con anterioridad el aprovechamiento de los descartes de pesca por parte de la pardela balear, su distribución no estuvo condicionada a la de los arrastreros. Ello sugiere que aunque los descartes de pesca pueden ser un recurso trófico importante durante algunos periodos, para completar los requerimientos energéticos, no influyen en su distribución, al menos a una gran escala espacial (de decenas a centenares de kilómetros). Por tanto, los descartes representarían, cuantitativamente, un recurso trófico importante aunque secundario al ser aprovechado de forma oportunista en ausencia de presas alternativas. Sin embargo, es difícil discernir cómo influyen los dos recursos tróficos (presas naturales y descartes pesqueros) en la distribución de la especie ya que las áreas productivas donde se alimentan las pardelas sustentan, al mismo tiempo, una importante flota de arrastre demersal.

11. En base a los resultados de la asociación al hábitat de la pardela balear, se propuso la delimitación de un área marina protegida donde las áreas de alta densidad (zonas de alimentación) definirían el núcleo, de especial interés para su conservación, donde se implementarían medidas de protección rigurosas (prohibición de tráfico de buques petroleros y adopción de medidas mitigadoras para evitar la captura incidental en artes de pesca): el área de influencia del río Ebro y el entorno del cabo de la Nao. Dentro del área de campeo de la especie se implementarían otras medidas de protección más difusas, como un programa de observadores en la flota palangrera y el estudio del impacto de un parque eólico marino en el delta del Ebro, para su conservación. La asociación de la pardela a hábitat oceanográficos dinámicos (sistemas frontales, plumas fluviales productivas) hace que delimitar los límites de estas áreas

marinas protegidas dinámicas sea más difícil que las basadas en características geográficas fijas y su extensión geográfica deba revisarse periódicamente.

12. Las futuras líneas de investigación, en cuanto a la asociación al hábitat marino de la pardela balear, deberían centrarse en obtener un entendimiento mecánico de la interacción, a pequeña escala, entre las condiciones oceanográficas, distribución de sus presas y los patrones de agregación y comportamiento de la pardela balear en las áreas de alimentación importantes en la plataforma continental Ibérica. Por otra parte, también es importante determinar la asociación al hábitat de la pardela balear cuando la especie se dispersa en el Mediterráneo occidental y en el Atlántico norte e identificar áreas marinas protegidas en dichas áreas biogeográficas.

13. La pesca de arrastre demersal genera una gran cantidad de descartes de la cual se aprovechan algunas poblaciones de aves marinas. En el archipiélago balear la actividad de descarte de la flota de arrastre se llevó a cabo, principalmente, en la costa suroeste de Mallorca donde se localiza el 12% de la población reproductora de aves marinas y el principal puerto de arrastre del archipiélago balear. El acoplamiento espacio-temporal de la abundancia de aves marinas asociada a los arrastreros dependía de la compleja interacción de las características de la pesca de arrastre. La abundancia de aves fue mayor en dos situaciones: (1) en el estrato costero y de plataforma durante el invierno y (2) cuando un gran número de arrastreros potentes se encontraba en la misma zona al inicio de la actividad de descarte.

14. La gaviota patiamarilla *Larus michaellis* y la pardela cenicienta *Calonectris diomedea* fueron las especies reproductoras más comunes, al tiempo que las más abundantes detrás de los arrastreros. La abundancia temporal de esta última parece estar asociada al inicio y final del periodo reproductor. Las otras cuatro especies reproductoras en el archipiélago balear realizaron un menor uso de los descartes. En el caso de la pardela balear, las abundancias fueron sorprendentemente menores en el archipiélago que en el delta del Ebro, probablemente debido a diferencias en el número de arrastreros y en la cantidad y composición del descarte en ambas zonas. La presencia de cormorán moñudo *Phalacrocorax aristotelis desmarestii*, tanto de adultos como de inmaduros, detrás de los arrastreros fue mayor que en años previos, y hasta gran distancia de la costa (hasta 20 km). La gaviota de Audouin *Larus audouinii* fue menos abundante que

en el delta del Ebro probablemente, como resultado de la competencia con la gaviota patiamarilla (la relación gaviota patiamarilla/gaviota de Audouin es mucho mayor en el archipiélago).

15. Durante el periodo de estudio se descartaron aproximadamente 1050 toneladas anualmente en Mallorca. El estrato de pesca influyó en la cantidad de captura descartada y desembarcada, así como en la proporción de descarte (DR), probablemente relacionado con la situación geográfica, características geomorfológicas y batimétricas de las pescas. Existió una fuerte asociación positiva entre la cantidad de captura descartada y desembarcada, en promedio los descartes correspondían al 74% de las capturas desembarcadas (rango: 9 - 600%). La composición de descarte se caracterizó principalmente por una mayor proporción de peces, seguida de crustáceos, moluscos y otros.

16. La comparación de las abundancias de aves registradas asociadas a los arrastreros en el presente estudio con años previos sugiere que los descartes representan una pequeña parte de los requerimientos energéticos de la comunidad de aves marinas en el archipiélago balear, por lo que ésta no hace un uso extensivo de este recurso trófico.

17. En los últimos años, se está promoviendo una gestión pesquera que priorice el ecosistema en su conjunto, más que las especies objetivo, a través de una explotación sostenible de los recursos vivos marinos. Desde el punto de vista de las aves marinas las regulaciones pesqueras, como la reducción de los descartes, requieren (1) primero de una gestión preventiva considerando las conexiones ecológicas entre la comunidad de aves marinas y el hábitat. En ausencia de descartes pesqueros, las poblaciones que hacen un uso extensivo de este recurso trófico pueden verse afectadas, reduciendo el éxito reproductor y la supervivencia adulta, y así afectando, al mismo tiempo, a la estructura de la comunidad. A largo plazo, la reducción de los descartes sin duda beneficiará la restauración del ecosistema en su conjunto y ésto, a su vez, a las poblaciones de aves marinas. (2) Segundo, la gestión pesquera debe evitar el agotamiento localizado de pequeños peces pelágicos (principal presa de las aves marinas en el Mediterráneo) en áreas de alimentación importantes. La depredación sobre las especies más vulnerables podría ser especialmente dramática si la reducción de los descartes se solapa con una reducción de la disponibilidad de presas naturales.

Por tanto, predecir la respuesta de las comunidades de aves marinas a los cambios en las regulaciones pesqueras es complicado y requiere de largas series temporales de datos para discernir entre los efectos de éstos y variaciones ambientales en las poblaciones de aves marinas, y más aún cuando las poblaciones están organizadas en metapoblaciones, ya que las acciones locales necesitan considerar el efecto global. Y por último, (3) las zonas de gran biodiversidad, como los frentes oceanográficos, deberían de protegerse de manera especial debido a su alto valor de conservación.

18. En definitiva, la pardela balear necesita urgentemente de una gestión pesquera sostenible y respetuosa con el ecosistema marino donde se proteja la propia especie y su papel ecológico por varias razones. Primero, porque existen evidencias de que la baja supervivencia adulta pueda deberse a una gran mortalidad en artes de pesca (palangre principalmente). Segundo, porque la ecología trófica de la especie está estrechamente relacionada con la distribución de sus presas pelágicas y disponibilidad de descartes de pesca, las cuales afectan a su productividad y distribución en el Mediterráneo occidental. Además, está asociada a frentes oceanográficos (áreas de alta biodiversidad) en la plataforma Ibérica donde la implementación de un área marina protegida ayudaría a salvaguardar esta especie endémica del archipiélago balear. Sin embargo, el establecimiento de esta área marina protegida puede ser polémico y sufrir grandes presiones socio-políticas ya que esta región sustenta, al mismo tiempo, el mayor caladero de pesca del Mediterráneo occidental.

Resumen

Resumen

Introducción

Actualmente, todos los ecosistemas naturales en el planeta han sido alterados por el ser humano, y los océanos no son ajenos a ello (Meffe y Carroll 1994, Stocker y Raible 2005, Hendriks et al. 2006, Malcolm et al. 2006). Aunque su vasta riqueza esté aún por descubrirse, los cambios ambientales producidos por el ser humano ya han tenido un efecto negativo sobre el funcionamiento de los ecosistemas y la dinámica de población de muchos organismos marinos (Botsford et al. 1997, Worm et al. 2005, UNEP 2006). Estos impactos antropogénicos están provocando cambios importantes en las densidades poblacionales de las aves marinas como resultado de la interacción de diversos factores, afectando sus rasgos de historia de vida y sus dinámicas poblacionales, así como la estructura de las comunidades (ver revisión en Montevecchi 2002, Furness 2003, Lewison et al. 2004, Votier et al. 2004). Las aves marinas son organismos longevos caracterizados por una alta supervivencia y bajas tasas de reproducción y su historia de vida específica hace que sean organismos vulnerables al no poder responder de forma rápida a la reducción de sus densidades poblacionales, debido a sus altos tiempos de generación (Croxall y Rothery 1991, Weimerskirch 2002). La supervivencia adulta es el parámetro demográfico que más influye en la tasa de crecimiento poblacional de un organismo longevo a nivel

local, mientras que la emigración y la inmigración lo son a nivel metapoblacional (Croxall y Rothery 1991, Oro 2003).

Una de las aves marinas más amenazadas del Mediterráneo es la pardela balear *Puffinus mauretanicus* (Arcos y Oro 2004, BirdLife International 2006). Por primera vez, se realiza una diagnosis de conservación para conocer la dinámica de población de la especie y conocer los parámetros demográficos que están afectando negativamente a la tasa de crecimiento poblacional de la pardela balear (Norris 2004). Así, se ha estimado la supervivencia adulta, rasgo fundamental para la conservación de una especie de larga vida (Croxall y Rothery 1991), y se ha realizado un análisis de viabilidad de la población estimando las probabilidades de extinción y tiempo de vida medio (capítulo 2). El éxito reproductor también puede ser otro rasgo de historia de vida fundamental para la conservación de la especie (Weimerskirch 2002), por lo que se ha estimado la contribución de cada uno de estos dos rasgos de historia de vida a la viabilidad de la especie, así como la influencia de las condiciones ambientales (recursos tróficos como los pequeños peces pelágicos y los descartes de pesca) sobre la productividad (capítulo 3).

La comparación de rasgos de historia de vida entre poblaciones locales de la misma especie puede darnos una idea de las adaptaciones a las condiciones ecológicas específicas en cada una de ellas (Weimerskirch 2002), y su variabilidad podría ser un mecanismo evolutivo para evitar la sincronización de las fluctuaciones poblacionales, importante para impedir la extinción simultánea de poblaciones locales y con consecuencias fatales para su conservación. Para ello, se realiza un análisis espacio-temporal de parámetros reproductores como el éxito de eclosión y reproductor (capítulo 3) y tamaño de huevo (capítulo 4) entre diferentes poblaciones locales de pardela balear. En el caso del tamaño del huevo, se analizan factores potenciales que podrían influir en su variabilidad, como el tamaño corporal y la ecología trófica.

El estudio de la ecología de la especie en las zonas de alimentación es imprescindible para conocer con detalle las amenazas a las que se enfrenta (Anderson et al. 2003, James et al. 2005), sobre todo en un contexto altamente modificado por el ser humano como es el Mediterráneo occidental, durante un periodo del ciclo reproductor crítico como es el de crecimiento de los pollos (capítulo 5). La identificación, a través de modelos, de las variables que se correlacionan con el hábitat nos puede ayudar a predecir la distribución de esta especie críticamente amenazada y la magnitud de determinados impactos. Así, se utiliza la asociación al hábitat de la especie para

delimitar el área de campeo e identificar áreas de alimentación importantes, caracterizados por altas densidades, y predecir aquellas áreas marinas importantes para la conservación de la especie. Al mismo tiempo, se sabe que la pardela balear se aprovecha de los descartes de pesca en zonas distantes a las colonias de cría, como el delta del Ebro, (Arcos y Oro 2002) aunque no se conoce con exactitud el grado de aprovechamiento de este recurso en el archipiélago balear. Por tanto, se estudia la explotación de este recurso por parte de la comunidad de aves marinas del archipiélago balear, haciendo especial hincapié en la especie de estudio, a lo largo de todo el año y caracterizando los patrones de distribución y estructura de la comunidad de aves marinas desde una perspectiva global, en relación a variables relacionadas con la pesquería, y otra más específica que permite explorar patrones espacio-temporales. Finalmente, se realizan recomendaciones para integrar la interacción de las aves marinas con la pesca en el manejo de las pesquerías del Mediterráneo occidental, ya que éstas no sólo afectan a los stocks de peces sino también al ecosistema marino en su conjunto.

Resultados y Conclusiones

Un estudio demográfico detallado permitió determinar los parámetros demográficos que afectan negativamente a la tasa de crecimiento poblacional: la mortalidad adulta es la principal causa del declive poblacional (capítulo 2), mientras que los valores actuales de éxito reproductor (otro parámetro demográfico de potencial relevancia para la conservación de la pardela balear) y los factores que pueden influir en este parámetro (la disponibilidad de alimento) no son cuestiones preocupantes para la conservación de la especie, al menos en la actualidad y en las colonias de estudio (capítulo 3). La baja supervivencia adulta (0.780, SE = 0.020, 95% intervalo de confianza: 0.739-0.816) en colonias libre de depredadores de aves adultas sugería que, a parte de los depredadores, podría haber otras fuentes de mortalidad (artes de pesca, por ejemplo) (capítulo 2). El análisis de viabilidad de la población mostró que las probabilidades de extinción de la pardela balear son extremadamente altas (40.4 años, SE = 0.2, de tiempo medio de extinción y una disminución anual de un 7.4% de la tasa de crecimiento medio), clasificando a la especie en Peligro Crítico, según criterios de IUCN (Unión Internacional para la Conservación de la Naturaleza), la categoría de amenaza mayor para una especie no extinta en libertad (capítulo 2).

La productividad varió con los años y de manera similar en todas las colonias. Así, la disponibilidad de alimento, tanto de pequeños peces pelágicos como de descartes de la pesca de arrastre en la zona del delta del Ebro (principal zona de alimentación de la pardela balear), también influyó positivamente en la productividad. La variabilidad temporal de la productividad probablemente fue debida a los cambios anuales en la disponibilidad de pequeños peces pelágicos, que exhiben estocasticidad temporal, y descartes pesqueros (capítulo 3).

En cuanto al tamaño del huevo, se vio que este rasgo de historia de vida variaba entre poblaciones locales vecinas y no con los años, resultado inesperado a una escala espacial tan pequeña, ya que las colonias de estudio presentaban características ecológicas similares (capítulo 4). La variabilidad espacial en el tamaño del huevo podría explicarse parcialmente por diferencias interpoblacionales en el nivel trófico en el que se alimentan los adultos reproductores (valores promedio del isótopo $\delta^{15}\text{N}$). Tanto las diferencias interpoblacionales en el nivel trófico como en el tamaño del huevo mostraron el mismo patrón general, es decir, los huevos más grandes estarían relacionados con valores promedio de $\delta^{15}\text{N}$ mayores, sugiriendo una conexión nutricional entre ambos. Al mismo tiempo los dos factores, el nivel trófico y el tamaño del huevo podrían estar influenciados por el tamaño corporal parental (potencial indicador de un componente genético) puesto que éste podría mediar en la selección de las presas y condicionar el tamaño del huevo.

Para la conservación de especies amenazadas es de vital importancia conocer su asociación al hábitat e implementar medidas de conservación como las áreas marinas protegidas (capítulo 5). En base a los resultados de la asociación al hábitat de la pardela balear se propuso la delimitación de un área marina protegida donde las áreas de alta densidad (zonas de alimentación, muda y corredores de migración entre otros) definirían el núcleo, de especial interés para su conservación, donde se implementarían medidas de protección rigurosas: el área de influencia del río Ebro y el entorno del cabo de la Nao. Dentro del área de campeo de la especie (aguas costeras de la plataforma continental Ibérica caracterizadas por sistemas frontales en áreas cercanas a las colonias de cría, entre 42.3° y 38.0°N), se implementarían otras medidas de protección más difusas (programa de observadores en la flota palangrera y el estudio del impacto de un parque eólico marino en el delta del Ebro) para su conservación. La variabilidad interanual de los patrones de distribución observados

probablemente era respuesta a diversos procesos oceanográficos como plumas fluviales, fenómenos de mesoescala y cañones submarinos que incrementan la productividad en zonas localizadas y agregan las presas (capítulo 5). Los modelos predictivos resultaron una herramienta útil, y con un gran potencial, para determinar la asociación al hábitat de la pardela balear tanto en áreas muestreadas como no muestreadas, ya que el hábitat oceanográfico predicho por el modelo final concordó con el área de campeo descrito a partir de los datos observados, e incluso identificó las aguas cercanas a las colonias de cría en las islas Baleares como hábitat adecuado para la especie (capítulo 5).

En el archipiélago balear la actividad de descarte de la flota de arrastre se llevó a cabo a lo largo de todo el año, principalmente, en la costa suroeste de Mallorca donde se localiza el 12% de la población reproductora de aves marinas y el principal puerto de arrastre del archipiélago balear. El acoplamiento espacio-temporal de la abundancia de aves marinas asociada a los arrastreros dependía de la compleja interacción de las características de la pesca de arrastre. La abundancia de aves fue mayor en dos situaciones: (1) en el estrato costero y de plataforma durante el invierno y (2) cuando un gran número de arrastreros potentes se encontraba en la misma zona al inicio de la actividad de descarte. Existió una fuerte asociación positiva entre la cantidad de captura descartada y desembarcada, en promedio los descartes correspondían al 74% de las capturas desembarcadas (rango: 9 - 600%) (principalmente peces seguido de crustáceos, moluscos y otros). La comparación de la asociación de las aves marinas con los barcos de arrastre en el archipiélago balear en años anteriores (Oro y Ruiz 1997) sugiere que los descartes representan una pequeña parte de los requerimientos energéticos de la comunidad de aves marinas en el archipiélago balear, por lo que ésta no utiliza de forma extensiva este recurso trófico.

En definitiva, la pardela balear necesita urgentemente de una gestión pesquera sostenible y respetuosa con el ecosistema marino donde se proteja la propia especie y su papel ecológico por varias razones. Primero, porque existen evidencias de que la baja supervivencia adulta pueda deberse a una gran mortalidad en artes de pesca (palangre principalmente). Segundo, porque la ecología trófica de la especie está estrechamente relacionada con la distribución de sus presas pelágicas y disponibilidad de descartes de pesca, las cuales afectan a su productividad y distribución en el Mediterráneo occidental. Además, está asociada a frentes oceanográficos (áreas de alta biodiversidad) en la plataforma Ibérica donde la implementación de un área marina

protegida ayudaría a salvaguardar esta especie endémica del archipiélago balear. Sin embargo, el establecimiento de esta área marina protegida puede ser polémico y estar bajo grandes presiones socio-políticas ya que esta región sustenta, al mismo tiempo, el mayor caladero de pesca del Mediterráneo occidental.

Referencias

- Anderson DJ, Huyvaert KP, Wood DR, Gillikin CL, Frost BJ, Mouritsen H (2003) At-sea distribution of Waved Albatrosses and the Galápagos Marine Reserve. *Biological Conservation* 110:367-373
- Arcos JM, Oro D (2002) Significance of fisheries discards for a threatened Mediterranean seabird, the Balearic shearwater *Puffinus mauretanicus*. *Marine Ecology Progress Series* 239:209-220
- Arcos JM, Oro D (2004) Pardela Balear, *Puffinus mauretanicus*. In: Madroño A, González C, Atienza JC (Eds) Libro Rojo de las Aves de España. Dirección General para la Biodiversidad- SEO/BirdLife, Madrid, pp. 46-50
- BirdLife International (2006) Species factsheet: *Puffinus mauretanicus*. Downloaded from <http://www.birdlife.org> on 14/7/2006.
- Botsford LW, Castilla JC, Peterson CH (1997) The management of fisheries and marine ecosystems. *Science* 277:509-515
- Croxall JP, Rothery P (1991) Population regulation of seabirds: implications of their demography for conservation. In: Perrins CM, Lebreton J-D, Hirons GJM (Eds), *Bird Population Studies, Relevance to Conservation and Management*. Oxford University Press, Oxford, pp. 272-296
- Earn DJD, Levin SA, Rohani P (2000) Coherence and conservation. *Science* 290:1360-1364
- Furness RW (2003) Impacts of fisheries on seabird communities. *Scientia Marina* 67S2: 33-45
- Hendriks IE, Duarte CM, Carlo HR (2006) Biodiversity research still grounded. *Science* 312:1715
- James MC, Ottensmeyer CA, Myers RA (2005) Identification of high-use habitat and threats to leatherback sea turtles in northern waters: new directions for conservation. *Ecology Letters* 8:195-201

-
- Lewison RL, Crowder LB, Read AJ, Freeman SA (2004) Understanding impacts of fisheries bycatch on marine megafauna. *Trends in Ecology and Evolution* 19:598-604
- Malcolm JR, Liu CR, Neilson RP, Hansen L, Hannah L (2006) Global warming and extinctions of endemic species from biodiversity hotspots. *Conservation Biology* 20:538-548
- Meffe GK, Carroll RC (1994) *Principles of Conservation Biology*, 2nd ed. Sinauer Associates, Sunderland, MA
- Montevicchi WA (2002) Interactions between fisheries and seabirds. In: Schreiber EA, Burger J (Eds) *Biology of Marine Birds*. CRC Press, Boca Raton, pp. 527-557
- Norris K (2004) Managing threatened species: the ecological toolbox, evolutionary theory and declining-population paradigm. *Journal of Applied Ecology* 41:413-426
- Oro D (2003) Managing seabird metapopulations in the Mediterranean: constraints and challenges. *Scientia Marina* 67S2:13-22
- Oro D, Ruiz X (1997) Exploitation of trawler discards by breeding seabirds in the north-western Mediterranean: differences between the Ebro Delta and the Balearic Islands areas. *ICES Journal of Marine Science* 54:695-707
- Stocker TF, Raible CC (2005) Climate change: water cycle shifts gear. *Nature* 434:830-833
- UNEP (2006) *Marine and coastal ecosystems and human wellbeing: A synthesis report based on the findings of the Millennium Ecosystem Assessment*. (available at <http://www.maweb.org/en/index.aspx>)
- Votier SC, Furness RW, Bearhop S, Crane JE, Caldow RWG, Catry P, Ensor K, Hamer KC, Hudson AV, Kalmbach E, Klomp NI, Pfeiffer S, Phillips RA, Prieto I, Thompson DR (2004) Changes in fisheries discard rates and seabird communities. *Nature* 427:727-730
- Weimerskirch H (2002) Seabird demography and its relationship with the marine environment. In: Schreiber EA, Burger J (Eds) *Biology of Marine Birds*. CRC Press, Boca Raton, pp. 115-135
- Worm B, Sandow M, Oschlies A, Lotze HK, Myers RA (2005) Global patterns of predator diversity in the open oceans. *Science* 309:1365-1369

