TESIS DOCTORAL

Enrique de la Montaña Andrés
2009

DIVERSIDAD DE VERTEBRADOS TERRESTRES EN ESPAÑA

Identificación de áreas para la conservación, mitigación de impactos y gestión del monte mediterráneo

Universidad de Alcalá
Departamento de Ecología

DIVERSIDAD DE VERTEBRADOS TERRESTRES EN ESPAÑA

Identificación de áreas para la conservación, mitigación de impactos y gestión del monte mediterráneo

Enrique de la Montaña Andrés

A efectos bibliográficos la obra debe citarse como sigue:

De la Montaña, E. 2009. Diversidad de vertebrados terrestres en España. Identificación de áreas para la conservación, mitigación de impactos y gestión del monte mediterráneo. Tesis Doctoral, Universidad de Alcalá, Alcalá de Henares (España).

Fotografías: Enrique de la Montaña

Maquetación: Enrique de la Montaña

Impreso en papel libre de cloro por: Imprenta de la U.A.H.

Universidad de Alcalá

DEPARTAMENTO DE ECOLOGÍA

Diversidad de vertebrados terrestres en España. Identificación de áreas para la conservación, mitigación de impactos y gestión del monte mediterráneo

Memoria presentada para optar al grado de
Doctor por la Universidad de Alcalá

Enrique de la Montaña Andrés

Alcalá de Henares, Febrero de 2009

DEPARTAMENTO DE ECOLOGÍA

ECOLOGíA
Campus Universitario
Ctra. Madrid-Barcelona, Km 33,600
E-28871 Alcalá de Henares, Madrid. Spain
Telf. 0034 918854927
Fax 0034 918854929
e-mail: ecologia@uah.es

José María Rey Benayas, Catedrático de Ecología de la Universidad de Alcalá,

Hace constar:

Que el trabajo descrito en la presente memoria, titulado **"Diversidad de vertebrados terrestres en España. Identificación de áreas para la conservación, mitigación de impactos y gestión del monte mediterráneo"**, ha sido realizado bajo su dirección por D. Enrique de la Montaña Andrés en el Departamento de Ecología de la Universidad de Alcalá, y reúne todos los requisitos necesarios para su aprobación como Tesis Doctoral.

Alcalá de Henares, a 9 de febrero de dos mil nueve.

Dr. José María Rey Benayas
DIRECTOR DE LA TESIS

DEPARTAMENTO DE ECOLOGÍA

ECOLOGíA
Campus Universitario
Ctra. Madrid-Barcelona, Km 33,600
E-28871 Alcalá de Henares, Madrid. Spain
Telf. 0034 918854927
Fax 0034 918854929
e-mail: ecologia@uah.es

Miguel Ángel Rodríguez Fernández, Profesor Titular y Director del Departamento de Ecología de la Universidad de Alcalá,

Hace constar:

Que el trabajo descrito en la presente memoria, titulado **"Diversidad de vertebrados terrestres en España. Identificación de áreas para la conservación, mitigación de impactos y gestión del monte mediterráneo"**, ha sido realizado por D. Enrique de la Montaña Andrés dentro del Programa de Doctorado Cambio Global y Desarrollo Sostenible adscrita al Departamento de Ecología de la Universidad de Alcalá, y reúne todos los requisitos necesarios para su aprobación como Tesis Doctoral.

Alcalá de Henares, a 9 de febrero de dos mil nueve.

Dr. Miguel Ángel Rodríguez Fernández
DIRECTOR DEL DEPARTAMENTO

A mis padres, por supuesto,
con eterno cariño y gratitud.

"El secreto de la felicidad no es hacer siempre lo que se quiere,
sino querer siempre lo que se hace"
León Tolstoi

Agradecimientos

Son muchas las personas que han hecho posible que llegase este momento. Son muchas las cosas que han pasado en todo ese tiempo. Todo forma un conjunto de recuerdos, de momentos especiales por los que quiero dar las gracias, no sólo porque me han animado a llegar a puerto sino porque me han hecho feliz en el trayecto, que es sin duda lo más importante.

El comienzo de todo esto fue un día nevado de marzo. Porque desde ese instante me empezaron a inculcar el respeto a todas las cosas y la pasión por conocer allá lejos. Gracias infinitas a mis padres, por criarme con mucho amor y libertad, aunque eso no suele ser fácil, ya que el amor tiende a ser posesivo y a mí siempre me han dejado volar. Pero mi destino era seguro, biólogo de nacimiento, a pesar de que a veces he desviado mi trayecto parando en distintos lugares. En ocasiones fue duro pero también una aventura inolvidable que me enseñó todo lo que soy.

Gracias a mi hermana Tamara, por ser como eres, por todo lo que haces por mí.

Gracias también a Peluso, a Colmillo Blanco, a los espalda plateada, al cañón del Río Dulce y al hermano lobo, a La Misión, a los pueblos indígenas, al "mira a tu alrededor", a la gran barrera, a la danza rosa en el Nakuru, a las aves del paraíso, al leopardo de las nieves, a los emperadores de la Antártida, a la vida secreta de las plantas, a las arribadas en cuarto creciente. Al secreto de las tortugas. A las pasiones y los sueños.

A José María, porque sin lugar a dudas, ha sido el mejor director de tesis que hubiera podido tener, y eso, más que una simple casualidad, se debe a toda la dedicación y paciencia que ha tenido conmigo. Ha sacado de mí todo lo que ha podido y siempre me ha tenido feliz.

Gracias a todos los propietarios de las fincas manchegas que me permitieron merodear por sus tierras contando pájaros. Más de dos años de muestreos con helados amaneceres y calores primaverales, muchas noches durmiendo en pleno campo, muchas tortillas de patata con el cariño de una madre. Disfruté la soledad metido en las entrañas de mi querido bosque mediterráneo.

También he de mencionar que este trabajo ha podido llevarse a cabo gracias a las becas predoctorales que me concedieron la Universidad de Alcalá y la Comunidad de Madrid. Además de contar con la financiación de varios proyectos de la CICYT (REN2000-745, CGL2004-00355/BOS, CGL2007-60533/BOS), de la Red de grupos de investigación REMEDINAL (proyecto S-0505/AMB/0355), del proyecto Forest de la Unión Europea (II-0411-FAFCD-FI-FC) y de la Fundación FIRE.

Para ser justos, hay que reconocer que un trocito muy grande de esta Tesis es de Ana y Lorenzo. Se podría escribir otro libro contando todo lo que ayudan para que salga el trabajo de los demás. No sólo sostienen este Departamento y lo hacen funcionar más eficientemente, sino que lo humanizan con su gesto siempre amable y dispuesto a solucionar todos los problemas que inventamos.

Gracias a Salva, al que quiero, que me ha contagiado su pasión, me ha ayudado cuando se lo he pedido y cuando no también. Hemos compartido muchos momentos increíbles y lo mejor es que seguro que seguiremos haciéndolo. A Nico, al que adoro, siempre atento y dispuesto a escucharnos con el corazón, motivándonos en tremendos valores. Lamento que nunca pude ser tu alumno en la carrera. A Juantxo, un ejemplo a seguir, me acogió y me llevó a las más altas cumbres, enseñándome lo duro y fascinante que es trabajar en lo que uno quiere de la manera que quiere. A Josa, gracias por todo el tiempo que dedicaste para mejorar nuestro trabajo y gracias por recordarme que no me olvide del continente helado. Gracias al resto de profesores que de alguna manera me han ayudado, Josevi, Tiscar, Pedro, Julio, Pilar, Asun y en especial a Miguel Ángel y Antonio, que me enseñaron como enamorarme de la Ecología cuando yo vagaba por las aulas.

También quiero agradecer a Luisma que me ayudara a dar sentido a un montón de datos. Contigo aprendí muchas cosas en muy poco tiempo, fue un placer, al igual que las interesantes charlas sobre fotografía. Otras excelentes colaboradoras fueron Irene, Inés y Ana, gracias por tantas horas de trabajo.

Muchas cosas buenas tiene haber estado tantos años por acá, una de las mejores es haber conocido a todos los becarios. Los primeros allá por el siglo XX, madre mía!!! Con ellos el Departamento empezó a crecer y a convertirse en un lugar no sólo de trabajo. Toño, Marcos, Álvaro, Pablo, Sara y René. Año tras año los becarios íbamos colonizando los despachos hasta que llegó el momento de máximo esplendor con la era del PEHE. Durante ese tiempo Carlota e Itzi, me conquistaron con su dulzura. Gracias por todo lo que me habéis aguantado y por toda la vida que me habéis enseñado, por vuestra compañía en el despacho, por los viajes de regreso, por las cañas y cenas. Sois dos joyas sin lugar a dudas. Gracias a Marta y a Lucia por tantos años, por tantas cosas. A Luis por regalarme un viaje austral y por tu ayuda, un brillante compañero y mejor persona, aunque en el fútbol debes mejorar. Gracias Noe por tu energía. Carlos, enhorabuena padrazo, gracias por tantos kilómetros de charla y música en busca de los sarrios y el orujo de Colungo. Y a Fabio mil gracias por ayudarnos a todo el departamento con esos temas informáticos que te encantan y que los demás desconocemos. Gracias a Alberto, Miki, Nacho, Dani, Maria José, Mariano, Oscar, Bárbara, Virginia, Margarita, Edu, Silvia, Irene, Rosa, Anabella, Liliana y a las nuevas caras que alegran los proyectos, en especial a Vir y Sara. Gracias a todos por hacer que sea un autentico placer levantarse por la mañana para ir a trabajar.

Gonzalo, Lorenzo y Luis, desde hace ya tiempo sois algo más que amigos. Siempre ahí, dispuestos a ayudar, siempre pendientes de lo que necesitan los demás. Gracias por vuestros consejos profesionales, así todo se hace más fácil. Siento ser un poco pesado y distraeros frecuentemente, que sepáis que siempre estaré en deuda con vosotros. Gracias por las cenas a la brasa y las comidas de cuchara, por los furanchos y los paseos por Somiedo, por las risas asfixiantes, por las aventuras en cualquier pico nevado o en cualquier lugar del mundo conocido o por conocer. Pero sobre todo gracias por vuestras orejas, tres pares auriculares que soportan mis historias sin rechistar. Impagable es lo que me habéis enseñado.

Gracias especiales a los que me han ayudado a descubrir el mundo, a crecer. En mis estancias en el extranjero he aprendido que soy muy afortunado por poder cumplir mis sueños, por eso no quiero dejar de soñar que se cumplan los sueños de los demás. A Pedrarias y su familia, que me regalaron tres meses de hamaca frente al Pacífico. Me trataron como si fuera su hijo, con ese toque Nica. Espero que pronto puedas defender tu tesis. Si pudiste con la revolución, esto no será problema. Viva León Jodido!!!. Risto, Velu, Elina, David y Sandra que me dieron su calor para que me sintiera como en casa en aquellos largos días del círculo polar, por tantas makrara. Amutete y Muchai, que me llevaron a muchos rincones del continente negro, durmiendo en fondas de dudoso honor y comiendo cabras Masai a orillas del Mara. Inolvidable black style. A Jay, que aceptó mi ayuda en Nunca Máis y me recompensó con una beca para el IBRRC, con su casa y con su coche. Me dio la oportunidad de abrir los ojos ante mis prejuicios y me enseñó que en el eje del mal hay mucha gente de bien. A todos los chilenos allá y a Chucu acá, porque nunca sabes donde habrá una sorpresa.

Y a mi querido Ecuador, que ha marcado mi vida con todo lo que me tenía reservado. A parte de su desbordante biodiversidad (necesitaba que saliera esta palabra en todas las partes de este libro), tuve la suerte de topar con SIMBIOE, con Irina, Roci, Delfi, Marisela y las Gagy. Pero especialmente con Ruth, Lina y Paúl, tres increíbles personas que forman parte de mi familia ecuatoriana. Me llenaron de ilusiones, me mostraron otro mundo. Debo agradecer también a los clanes de D. Guillermo Payaguaje y Aníbal Criollo y el resto de Secoyas y Cofán, que me acogieron y enseñaron cómo se vive con nobleza y humildad, mientras luchan por evitar que nuestro mundo incivilizado se los coma. Gracias a la fuerza de Cocaya y al espíritu de Masakay. Millones de gracias Ruth, de tu Gorrión abducido.

En los pasos finales de la tesis gracias a los que me habéis echado una mano, Vir, Luis, Lorenzo y Mariano. A Gonzalo porque sé que si hubieras estado aquí lo habrías hecho. A Javi, por documentarme para ilustrar esta memoria.

A Sandra, Ceci y Ana por supuesto. Lourdes y Elena con su Icia, que siempre tienen para mí un trocito de fiesta. Alfon, Cristi y el resto de hombres rana del Calypso. A Javi y Marta desde la carrera, y a Ruben desde la adolescencia, por los buenos ratos. Y a ti Rebe, por hacerme volar alto.

Recuerdos especiales al fútbol sala y fútbol siete, a los partidos de voley y torneos de raqueta, donde hacíamos mucho más que deporte. A las jornadas de cañas y tapas. A todas las sobremesa en la densa cafetería y a los pic-nic soleados. A las victorias y fracasos de los 101. A las fiestas en casa de Alberto y Carlota. A los cumpleaños de marzo. A los censos de Cuenca.

Y a ti, por todo lo que me has regalado, por hacerme un poquito bastante feliz y por lo que nos queda por VIVIR.

MUCHAS GRACIAS A TODOS, POR TODO.

Kique

Resumen

La degradación de los ecosistemas debida a las actividades humanas está provocando una pérdida de biodiversidad sin precedentes en la historia del planeta y, como consecuencia, una reducción de los bienes y servicios que proporcionan. Esta Tesis Doctoral tiene como objetivo principal la provisión de herramientas e información útiles para la conservación de la diversidad de vertebrados terrestres en España. En ella se investigan, a distintas escalas de trabajo, métodos para determinar la diversidad de estos vertebrados y su aplicación para la identificación de áreas de conservación, la mitigación de impactos ambientales de obras civiles y la gestión del monte mediterráneo. Para identificar las áreas relevantes de diversidad de anfibios, reptiles, aves y mamíferos en España continental e Islas Baleares se utilizaron cinco criterios: la riqueza de especies, la rareza de éstas, su grado de vulnerabilidad, un nuevo Índice Combinado de Biodiversidad de los tres criterios anteriores y el también original Índice Estandarizado de Biodiversidad (IEB). Este IEB permite establecer un único valor de biodiversidad no apriorístico para una combinación de grupos taxonómicos en cada unidad analítica del territorio estudiado. Con los mapas de áreas relevantes de diversidad generados, se realizó un análisis de huecos (*gap analysis*) en la Red de Espacios Naturales Protegidos y se evaluó de qué manera los grandes proyectos de obras civiles previstos a nivel nacional afectan a la herpetofauna. También se realizó un ejemplo de planificación sistemática de la conservación donde se identificaron las áreas necesarias para garantizar la conservación de los vertebrados en Castilla-La Mancha. Por último, se presenta un "experimento natural" para evaluar los efectos del aclaramiento del monte mediterráneo o resalveo sobre la comunidad de aves en la provincia de Ciudad Real. Los resultados indican que el Índice Combinado de Biodiversidad y la rareza de especies son los criterios que mejor representan la diversidad de vertebrados terrestres en el área de estudio. La riqueza de especies no es la manera más eficiente de maximizar la representación de la biodiversidad, a pesar de que es usado comúnmente. La escasa congruencia encontrada entre las áreas relevantes de diversidad identificadas según los distintos criterios y para los diferentes taxones causa dificultades para desarrollar estrategias de conservación a escalas amplias. La comparación de los mapas de áreas relevantes de diversidad con la Red Nacional de Espacios Naturales Protegidos mostró que existe un 30.8% de huecos de 50 x 50 km según el IEB. En Castilla-La Mancha, las áreas propuestas para formar parte de la Red Natura 2000, mejoraron considerablemente la representación dada por los espacios protegidos, pero tampoco incluyeron todas las áreas relevantes de diversidad. La comparación de los mapas de áreas relevantes de diversidad de herpetofauna con las infraestructuras planeadas en la Península Ibérica mostró una coincidencia moderada (35.4% para anfibios y 31.2% para reptiles). Los resalveos del monte mediterráneo denso aumentaron la diversidad estructural del hábitat y la riqueza local de especies de aves. Este tratamiento selvícola presenta unos valores de conservación y económicos añadidos porque también resultó beneficioso para las especies de aves amenazadas a nivel europeo y las especies cinegéticas. Los métodos y conclusiones presentados en esta Tesis proveen información de utilidad para la mejora de redes de áreas de conservación, la mitigación de impactos ambientales, el manejo forestal sostenible y la restauración de los ecosistemas.

Palabras clave: Agroecosistemas; Conectividad; Conservación; Espacios naturales protegidos; Especies amenazadas; Índice Combinado de Biodiversidad; Infraestructuras civiles; Masa corporal; Monte mediterráneo; Rareza de especies; Red Natura 2000; Región mediterránea; Resalveo; Riqueza de especies; Vertebrados; Vulnerabilidad

Abstract

Ecosystems degradation is causing a loss of biodiversity without precedent in the planet's history and, as a consequence, a reduction in the good and services that they provide. This doctoral thesis has the aim to provide a set of tools and information useful for conservation of terrestrial vertebrates diversity in Spain. Within it, research is carried out, at different working scales, on methods to determine diversity of these vertebrates and its application for the identification of conservation areas, mitigation of environmental impacts and Mediterranean woodland management. To identify the areas of high-value diversity of amphibians, reptiles, birds and mammals in continental Spain and Balearic Islands, five criteria has been used: species richness; species rarity; their degree of vulnerability; a new Combined Index of Biodiversity; and also the original Standardized Biodiversity Index (SBI). This SBI allows to establish a unique value of biodiversity for several taxa in each analytical unity of study area. With the maps of high-value diversity created, a gap analysis was carried out in the Protected Areas Network and it has been evaluated to see how large infrastructure projects planned at national level, affect the herpetofauna. As well, an example of systematic conservation planning has been done, to identify the required areas to guarantee the conservation of vertebrates in Castilla- La Mancha. Finally, a "natural experiment" is presented at regional scale to evaluate the effects that the silvicultural thinning of Mediterranean maquis called resalveo has over the bird community of Ciudad Real province. The results indicate that the Combined Index of Biodiversity and the rarity of species are the two criteria that best represent the biodiversity of terrestrial vertebrates in the study area. Although the species richness is commonly used, it is not the most efficient way to maximize the representation of biodiversity. The scarce congruence found in the areas of high-value diversity identified according to different criteria and taxa, causes difficulties to develop conservation strategies at broad scales. The comparison between areas of high-value diversity with the Protected Areas Network shows that exist 30.8% of gaps of 50 x 50 km according SBI. In Castilla-La Mancha, the suggested areas to form part of the Natura 2000 Network, improves considerably the representation given by the protected areas although it doesn't include all the high-value diversity areas. The comparison between areas of high-value herpetofauna diversity with the planned infrastructures in the Iberian Peninsula shows a moderate coincidence (35.4% for amphibians and 31.2% for reptiles). Thinning of dense Mediterranean woodland increased the structural diversity of habitat and the local richness of bird species. The conservation and economical value of this forestry management is increased since it is also beneficial towards threatened bird species in Europe and gamebird species. The methods and conclusions in the thesis provide useful information for the improvement of conservation area networks, the mitigation of environmental impacts, the sustainable forestry management and the restoration of ecosystems.

Keywords: Agroecosystems; Body mass; Connectivity; Conservation; Combined Index of Biodiversity; Infrastructures; Mediterranean region; Natura 2000 Network; Protected areas; Species rarity; Species richness; Thinning, Threatened species; Vertebrates, Vulnerability

Índice

Capítulo 1

“El sentimiento más importante que el hombre puede experimentar es su respeto al misterio; éste es la fuente del arte y la ciencia. Quien no puede contemplar el mundo con asombro, es que tiene los ojos cerrados”

Albert Einstein

La Historia de la Vida. Ilustración de Sandra Shomaker

Capítulo 1

Introducción general

Conocer para conservar. Estimar los atributos naturales de los ecosistemas es la base para desarrollar cualquier iniciativa de conservación de la Naturaleza, que quiera ser rentable ecológica y económicamente (James *et al*. 1999). Crear una sólida base de conocimiento será útil para, por ejemplo, el diseño y la mejora de redes de áreas protegidas, el aprovechamiento racional de los recursos naturales, la restauración de los ecosistemas o la mitigación de impactos ambientales.

La necesidad de preservar la Naturaleza se reconoce desde antiguo. Muchas culturas poseen creencias religiosas y filosóficas sobre el valor de la protección de distintas especies de fauna y flora y la necesidad de vivir en armonía con la Naturaleza (Hargrove 1989, Callicott 1994). Más actualmente, la hipótesis de Gaia propuesta por J. Lovelock (1979) representa una perspectiva similar al considerar a la Tierra como un "superorganismo" donde sus componentes biológico, físico y químico interaccionan para mantenerle vivo. El actual paradigma del desarrollo sostenible propugna la utilización de los recursos naturales sin dañar las comunidades biológicas ni comprometer las necesidades de generaciones futuras (Lubchenco *et al*. 1991).

El termino biodiversidad es un neologismo que nace como una contracción de diversidad biológica, en el Simposium Nacional sobre Biodiversidad celebrado en Washington en 1986. La publicación del libro BioDiversity (Wilson 1988), con las conclusiones de dicho simposium, presagió la popularidad de este concepto. La biodiversidad es interpretada por Wilson (1988) como un concepto holístico, que incluye la totalidad de los diferentes organismos, los genes que contienen y los ecosistemas que forman. La definición que propone la Convención sobre la Diversidad Biológica (CBD) es "*the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems*" (UNCED 1992). Pero la diversidad biológica ha sido fuente de asombro y curiosidad científica desde Darwin (1859) y Wallace (1876). La aceleración de los efectos de las actividades humanas en nuestro planeta renueva el interés de conocer como la biodiversidad de un nivel trófico o gremio, afecta a la dinámica y funcionalidad de poblaciones, comunidades y ecosistemas. Dicho interés se hace patente con la publicación del primer libro sobre los efectos de la biodiversidad en los ecosistemas por Schulze & Mooney en 1993. La diversidad de especies influye en el funcionamiento de los ecosistemas mediante la suma de los efectos de distintos componentes: número de especies presentes (riqueza), abundancia relativa (equitatividad), identidad de las especies (composición), las interacciones entre especies y la variación temporal y espacial de estas propiedades (ver revisión en Chapin *et al*. 2000). De este modo, la diversidad de especies repercute en la resistencia y resiliencia de los ecosistemas a cambios ambientales (Figura 1.1).

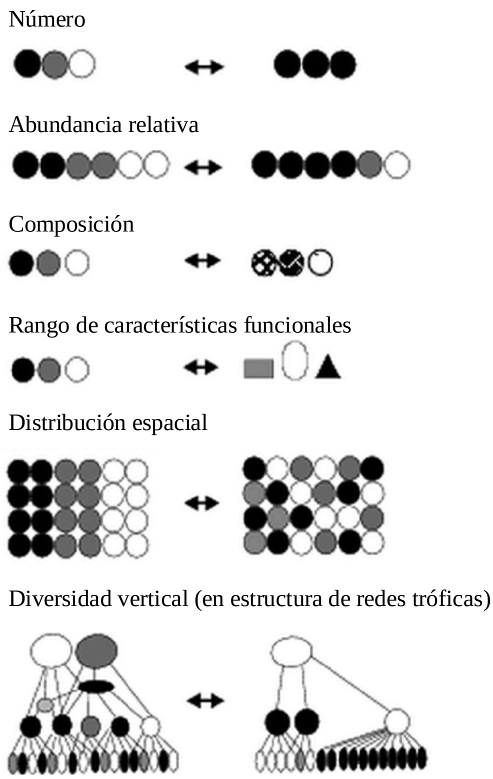


Figura 1.1. Representación de los diferentes componentes de la biodiversidad. Estos componentes pueden ser afectados por la intervención humana (flechas), y los cambios tienen repercusiones en la propiedades y servicios de los ecosistemas. Los símbolos representan individuos o unidades de biomasa. Símbolos de diferentes colores representan diferentes genotipos, fenotipos, o especies. Fuente: Díaz *et al*. (2006).

Las investigaciones desarrolladas en las últimas dos décadas sobre los efectos de la biodiversidad en los ecosistemas han mostrado que, en general, a mayor diversidad mayor estabilidad ecológica, mayor productividad, mayor retención de nutrientes en los ecosistemas y mayor resistencia frente a la invasión de especies exóticas (Pimm 1991, Tilman 1999, McCann 2000). Estos beneficios ecológicos han dado lugar a distintos argumentos de conservación, que se apoyan en el hecho de que la biodiversidad promueve los servicios ecosistémicos que garantizan el bienestar de la humanidad (Daily 1997, Loreau *et al*. 2002, Millenium Ecosystems Assessment 2005). La simplificación de los hábitats y la perdida de diversidad disminuirán la capacidad de los ecosistemas de proveer a la sociedad bienes y servicios esenciales de manera estable y sostenible (Tilman 2000, Balmford *et al*. 2002, Díaz *et al*. 2006). Reemplazar las fuentes de alimentos, combustibles, materiales industriales, medicamentos o recursos genéticos tendría unos elevados costes económicos (Costanza *et al*. 1997, Pimentel *et al*. 1997) y sociales, con inevitables consecuencias para la calidad de vida de la humanidad.

Estos argumentos pragmáticos atribuyen un valor económico a la biodiversidad mediante métodos que tienen en cuenta el valor de uso directo (productos recolectados) e indirecto (soporte y regulación de los ecosistemas) (McNeely *et al*. 1990, Barbier *et al*. 1994). Además, consideran un valor de opción basado en el potencial de proporcionar un beneficio en el futuro y el valor de lo desconocido (IUCN 1980), referido a especies conocidas pero también a valores desconocidos de especies desconocidas. A la biodiversidad también se le atribuye un valor de existencia basado en cuánto está la gente dispuesta a pagar para impedir que algún elemento de la biodiversidad desaparezca. Junto a estos argumentos existen otros de carácter ético, que afectan incluso a especies sin valor económico evidente y que por sí solos justifican los esfuerzos de conservación. Consideraciones tales como el derecho a existir, la responsabilidad hacia las generaciones futuras y valores espirituales y estéticos (Deane-Drummond 2004, Sarkar 2005). Por otro lado, existe el propio valor intrínseco conferido por la historia evolutiva y los papeles ecológicos únicos de las especies.

Tales argumentos se enfrentan con la actual crisis ambiental. Mientras las predicciones sobre el cambio climático empiezan a cumplirse (Houghton *et al*. 1992), la población humana crece exponencialmente y el constante desarrollo demandado por la humanidad exige el consumo de cada vez más recursos naturales. En todo el planeta las actividades humanas están transformando los sistemas naturales. Debido a ello las comunidades biológicas están siendo afectadas negativamente a causa de la destrucción y degradación de hábitats, la sobreexplotación de especies y al impacto de las especies invasoras (Heywood 1995, Chapin *et al*. 2000). Todo ello se ve potenciado por la desigual distribución de la riqueza en el mundo y la miseria de muchos de los países tropicales que poseen una biodiversidad relevante. La dominación humana de los ecosistemas de la Tierra (Vitousek 1997) está reduciendo alarmantemente la diversidad de especies y acelerando las tasas de extinción. En la actualidad estas tasas son del orden de 100 a 1000 veces mayores que durante el pasado geológico (Lawton & May 1995, Pimm *et al*. 1995). La biodiversidad de la Tierra está siendo destruida a un ritmo sin precedentes, de alcance indeterminado y consecuencias irreversibles para la vida en nuestro planeta. Si la tendencia actual continua, podemos esperar el fin de la Naturaleza tal como la conocemos.

Desde la década pasada se están empezando a dar algunos pasos positivos para tratar de solucionar la situación actual. En 1992, la I Convención sobre la Diversidad Biológica (UNCED 1992), firmada por 175 países, refleja el consenso global sobre la importancia de la biodiversidad para mantener los sistemas que hacen posible la vida sobre la Tierra. Más recientemente, en la Cumbre Mundial sobre Desarrollo Sostenible celebrada en Johannesburgo en 2002, los 190 países presentes se comprometieron a "…lograr, para el año 2010, una reducción significativa del ritmo actual de pérdida de diversidad biológica, a nivel mundial, regional y nacional…" (UNEP 2002). A nivel europeo, la Comisión de la Unión Europea aprobó en 1998 una estrategia en materia de biodiversidad y en 2001 un plan de acción. Para dar respuesta a esta crisis de biodiversidad, también se han generado múltiples investigaciones e iniciativas a diferentes escalas de trabajo, que estiman distintos componentes de la biodiversidad para priorizar áreas donde realizar tareas de conservación. Uno de los métodos más comunes es la identificación de *hotspots*, o áreas con excepcional biodiversidad y donde existe una elevada perdida de hábitat (Reid 1998, Myers *et al*. 2000, Sarkar *et al*. 2002). También existen trabajos que identifican las ecoregiones en crisis (Olson & Dinerstein 2002, Hoekstra *et al*. 2005), o que establecen las áreas prioritarias a escala global para la conservación de algún taxón concreto (Stattersfield *et al*. 1998). Otras investigaciones incluyen los costes económicos de la conservación dentro de sus algoritmos para determinar los sitios candidatos (Drechsler 2005, Strange *et al*. 2006, Wilson *et al*. 2006, Murdoch *et al*. 2007, Underwood *et al*. 2008).

El bioma mediterráneo está considerado como un punto clave para la conservación de la biodiversidad global (Myers *et al*. 2000, Brooks *et al*. 2006). Las regiones mediterráneas se caracterizan por tener una biodiversidad con elevada endemicidad (Cowling *et al*. 1996) y vulnerabilidad (Rundel 1998), que es susceptible de sufrir grandes cambios debido a su alta sensibilidad a los principales factores de cambio global en la biosfera (Sala *et al*. 2000). Con menos del 5% del bioma mediterráneo protegido a nivel mundial (Hoekstra *et al*. 2005), la selección eficiente de áreas protegidas adicionales es, sin duda, una tarea fundamental para alcanzar los objetivos de conservación global acordados en la VII Convención sobre la Diversidad Biológica (UNEP 2004). Pero fuera de las áreas protegidas, los ecosistemas manejados por el hombre también contribuyen a la conservación (Pimentel *et al*. 1992, Halladay & Gilmour 1995). En el futuro, estas áreas serán aun más importantes (Daily 2001) debido a la esperable merma de los ecosistemas libres de influencia humana. La cuenca mediterránea y gran parte del resto de Europa son el resultado de siglos de intervención humana (Blondel & Aronson 1999), y muchas especies silvestres se han adaptado a los diversos paisajes heterogéneos de origen antrópico. La conservación de la biodiversidad debe apoyarse en una adecuada gestión de los ecosistemas manejados por el hombre.

Natura 2000 (92/43/EEC) plantea una oportunidad, a nivel europeo, para compatibilizar la conservación y el desarrollo sostenible. Este tipo de políticas internacionales pueden ser efectivas para abordar temas de conservación en áreas geográficas grandes (Donald *et al*. 2007). En el caso de la actual Política Agraria Comunitaria, existe un vivo debate sobre la efectividad de las medidas agroambientales para producir beneficios ecológicos (Aebischer *et al*. 2000, Peach *et al*. 2001, Vickery *et al*. 2004), a pesar de que un número significativo de dichas medidas, están dirigidas específicamente a la conservación de la biodiversidad (Kleijn & Sutherland 2003). Mejorando su base de conocimiento y desarrollando objetivos cuantitativos adaptados a las peculiaridades locales, se puede conseguir que la PAC sea una importante herramienta para la conservación de la biodiversidad (Kleijn *et al*. 2006). Similares fundamentos deben considerarse en las políticas de pesca, forestal, hidráulica o energética, para desarrollar mecanismos de gestión sobre una sólida base científica que garantice su eficacia. Especialmente en aquellas actividades subvencionadas y/o destinadas a la conservación de la biodiversidad. Es el caso de los resalveos en masas arboladas de montes bajos de frondosas (Cuadro 1.1). Como ponen de manifiesto los textos de selvicultura (González-Molina 2005, Serrada 2008), estos trabajos se realizan para prevención de incendios forestales,

Cuadro 1.1: Resalveo

Se trata de un tratamiento selvícola que, por medio de una combinación de corta, poda y desbroce de distintas características, reducen la densidad de la masa forestal (nº de pies/ha) y la competencia para proporcionar mayor vigor a un número más o menos reducido de pies de cierto tamaño (resalvos). El más frecuente en la actualidad es el denominado resalveo de conversión (Imagen 1.1), que se orienta, como su nombre indica, a la conversión de masas de frondosas en monte bajo a monte alto. Consiste en la selección y poda de los resalvos de mayor tamaño y mejor porte, y corte de los sobrantes (Imagen 1.2), eliminando al menos el 50% de los pies presentes de más de 2 cm. de diámetro o un 30-35% del área basimétrica (Serrada *et al*. 2008). Es frecuente que este tratamiento vaya acompañado del desbroce y limpia manual de matorral. Cuando los resalveos se aplican en masas cuyo objetivo último es el adehesamiento, también se realizan podas de formación en los pies que se pretende que constituyan la masa final.

Imagen 1.1. Resalveo de conversión. La zona resalveada aparece en segundo plano detrás de las encinas de mayor porte. Al fondo se aprecia un área de monte bajo mediterráneo denso.

Imagen 1.2. Detalle de un resalvo seleccionado junto a los pies eliminados.

mejora y restauración de las cubiertas vegetales, aumento del aprovechamiento de los recursos del monte e incluso para la mejora del hábitat de especies amenazadas como el lince ibérico (*Lynx pardinus*) (San Miguel 2006).

Mediante la evaluación de distintos métodos para determinar la diversidad de vertebrados, esta Tesis Doctoral aporta información de utilidad para abordar las necesidades de la conservación de la biodiversidad mencionadas anteriormente. Cuando se habla de diversidad desde el punto de vista biológico, se suele hacer referencia a la relación entre el número de especies y la abundancia relativa de individuos en una comunidad biológica determinada. Los índices clásicos usados con mayor frecuencia para estimar esta diversidad son los de Simpson, Shannon-Weaver y Margalef (ver revisiones en Margalef 1974 o Magurran 1989). Debido a las dificultades que plantea calcular la abundancia de individuos, existen otros tipos de definiciones cuantitativas de la diversidad de especies, que se han desarrollado para poder comparar la diversidad global de diferentes comunidades a diferentes escalas geográficas. Son la diversidad α, la diversidad β y la diversidad γ. En esta Tesis Doctoral hemos utilizado la diversidad α en los trabajos de campo y la diversidad γ en los trabajos a mayor escala, pero en ambos casos nos referimos al número de especies presentes en un área geográfica determinada.

Existe una amplia literatura que utiliza la riqueza de especies en enfoques macroecológicos dirigidos a temas de conservación (Prendergast *et al*. 1993, Conroy & Noon 1996, Williams *et al*. 1996, Castro *et al*. 1997, Atauri & de Lucio 2001, Virolainen *et al*. 2001, Jetz *et al*. 2004, Rodrigues *et al*. 2004, Sergio *et al*. 2005, Jiguet & Julliard 2006, Salomón *et al*. 2006, Xu *et al*. 2008). En otras ocasiones se tienen en cuenta otras características de las especies presentes en una comunidad biológica determinada. Frecuentemente se consideran algunas de las medidas de rareza que responden a diferentes combinaciones de rango geográfico, abundancia local, especificidad de hábitat y ocupación de hábitat (Rabinowitz 1981, Rey Benayas 1999). Especialmente se refieren al tamaño del rango geográfico de las especies (Howard 1991, Prendergast *et al*. 1993, Williams *et al*. 1996, Castro *et al*. 1997, Jetz *et al*. 2004, Beazley *et al*. 2005, Jiguet & Julliard 2006) y al endemismo de las especies (Myers *et al*. 2000, Harris *et al*. 2005, Wilson *et al*. 2006, Xu *et al*. 2008). También es habitual considerar el grado de amenaza de las especies (Rodrigues *et al*. 2004, Sergio *et al*. 2005, Milner-Gulland *et al*. 2006), así como taxones indicadores (Faith & Walker 1996, Larsen *et al*. 2007) y especies emblemáticas (Sergio *et al*. 2005). Estos criterios también se usan, por ejemplo, para desarrollar algoritmos basados en los conceptos de complementaridad y de persistencia (Araujo & Williams 2000, Williams & Araujo 2002, Cabeza *et al*. 2004, Williams *et al*. 2004, Arponen *et al*. 2005, Salomón *et al*. 2006).

Debido a la baja congruencia detectada entre varios de estos criterios, aún existe una considerable controversia sobre qué medida utilizar y las consecuencias de aplicar diferentes medidas (Harcourt 2000, Mace *et al*. 2000, Fleishman *et al*. 2005, Orme *et al*. 2005, Grenyer *et al*. 2006, Lamoreux *et al*. 2006). Cada uno de estos criterios tiene ventajas e inconvenientes. Por lo tanto, es necesario investigar más su adecuación para alcanzar los distintos objetivos de conservación. También es importante desarrollar nuevos algoritmos o índices que apliquen simultáneamente diferentes criterios a distintos grupos taxonómicos, para cuantificar de una forma no apriorística la relevancia de la biodiversidad de una unidad del territorio.

A una escala local, también existen trabajos que estudian cómo la riqueza de especies y otros componentes de la diversidad se ven afectados por la gestión forestal en distintos ecosistemas (Sekercioglu 2002, Thompson *et al*. 2003, Campbell *et al*. 2007, Atwell *et al*. 2008). Sin embargo, no existe un adecuado conocimiento del efecto que tienen los resalveos del monte mediterráneo sobre la biodiversidad. Son necesarias investigaciones que aporten la información fundamental para desarrollar pautas de gestión que tengan en cuenta el funcionamiento de cada ecosistema, y conseguir que el aprovechamiento de los recursos naturales sea compatible con la conservación de la biodiversidad.

Estructura de la memoria doctoral

La presente memoria doctoral se ha organizado por capítulos en formato de artículos científicos. A este capítulo introductorio (**Capítulo 1**), le siguen los capítulos 2, 3, 4 y 5, que reproducen el contenido de artículos publicados o en revisión en diferentes revistas científicas, por lo que se presentan en inglés, con sus correspondientes secciones de introducción, material y métodos, resultados y discusión, todos ellos precedidos por un resumen en castellano. Por último, el capítulo 6 desarrolla unas consideraciones adicionales y el capítulo 7 presenta las conclusiones de esta Tesis Doctoral. Cada capítulo tiene su propia sección de referencias. A continuación se describe el contenido de los capítulos principales.

En el **Capítulo 2** (Rey Benayas, J.M., & de la Montaña, E. 2003. *Identifying areas of high-value vertebrate diversity for strengthening conservation. Biological Conservation* 114, 357-370) se definen las áreas relevantes de diversidad de vertebrados terrestres (anfibios, reptiles, aves y mamíferos) en España continental e Islas Baleares. Para ello se cuantifica en unidades analíticas de 50 x 50 km, la riqueza de especies, la rareza de éstas en términos de su distribución geográfica y su vulnerabilidad según su catalogación o no como especies amenazadas. También se proponen dos nuevos índices: el Índice Combinado de Biodiversidad, que integra los tres criterios descritos anteriormente para un grupo taxonómico y el Índice Estandarizado de Biodiversidad, que incluye los cuatro taxones juntos y permite establecer un valor de biodiversidad sintético en cada unidad analítica. Con los mapas de áreas relevantes de diversidad generados, se ha realizado un análisis de huecos (*gap analysis*) en la Red de Espacios Naturales Protegidos.

En el **Capítulo 3** (Rey Benayas, J.M., de la Montaña, E., Belliure, J. & Eekchout, X.R. 2006. *Identifying areas of high herpetofauna diversity that are threatened by planned infrastructure projects in Spain. Journal of Environmental Management* 79, 279-289) se identifican las áreas relevantes de diversidad de herpetofauna mediante el uso de los mismos criterios del capítulo anterior. Los análisis se realizan en celdas de 20 x 20 km, lo que permite evaluar la eficacia de los distintos criterios a una escala de trabajo de mayor detalle. Para garantizar la conservación de todas las especies de anfibios y reptiles es necesario identificar amenazas antes de que ocurran y así poder mitigar sus posibles impactos. Por ello, en este trabajo se evalúa de qué manera afectan a las áreas relevantes identificadas los grandes proyectos de infraestructuras previstos en España.

En el **Capítulo 4** (De la Montaña, E., Rey Benayas, J.M., Razola, I. & Vasques, A. En revisión. *Systematic conservation planning of vertebrate diversity. A case study in a Mediterranean European region. Conservation Biology*) se muestra un ejemplo de planificación sistemática de la conservación donde se identifican las áreas necesarias para garantizar la conservación de los vertebrados en Castilla-La Mancha. Se utiliza el Índice Estandarizado de Biodiversidad para identificar elementos especiales de conservación. Éstos se unieron a las áreas de conservación propuestas por la región para formar parte de la Red Natura 2000. Finalmente se seleccionaron áreas adicionales de conectividad que incluyeran los hábitats considerados poco representados por las áreas de conservación existentes.

En el **Capítulo 5** (De la Montaña, E., Rey Benayas, J.M. & Carrascal, L.M. 2006. *Response of bird communities to silvicultural thinning of Mediterranean maquis. Journal of Applied Ecology* 43, 651-659) se presenta un "experimento natural" a escala regional para evaluar los efectos del resalveo del monte mediterráneo sobre la comunidad de aves. El resalveo es un tipo de gestión forestal muy extendido que consiste en la eliminación de la mayoría de los matorrales y árboles pequeños, y la poda de los árboles más altos para producir masas forestales más abiertas. Conocer cuáles son las consecuencias de esta actividad es importante para la conservación de la biodiversidad a escala local. Se incluyen consideraciones sobre distintos gremios funcionales según el uso del hábitat y las preferencias tróficas, así como sobre grupos de especies de interés especial. En este capítulo se analiza la hipótesis de la existencia de un efecto alométrico en la masa corporal media de la comunidad de aves debido a la distinta complejidad estructural de los hábitats.

En el **Capítulo 6** se exponen unas consideraciones adicionales relativas a los problemas asociados a este tipo de trabajos de investigación. Se discuten aspectos no sólo desde el punto de vista teórico y metodológico, sino también se abordan ciertas dificultades para su aplicación práctica dentro del contexto socioeconómico actual.

En el **Capítulo 7** se relacionan las principales conclusiones de esta Tesis Doctoral.

Objetivos de la Tesis Doctoral

Esta Tesis Doctoral tiene el objetivo de proveer herramientas e información útiles para la conservación de la biodiversidad. En ella se investigan, a distintas escalas de trabajo, métodos para determinar la diversidad de vertebrados terrestres, y su aplicación para la identificación de áreas para la conservación, la mitigación de impactos ambientales y la gestión del monte mediterráneo en España.

Objetivos específicos

1. Identificar las áreas relevantes de diversidad de vertebrados terrestres a distintas escalas en España (Capítulos 2, 3 y 4).

2. Evaluar la eficacia de los distintos criterios utilizados para definir las áreas relevantes de diversidad. Investigar la hipótesis de que el Índice Combinado de Biodiversidad es mejor método para identificar áreas con características relevantes de biodiversidad que los comúnmente usados criterios de riqueza, rareza y grado de amenazada de las especies (Capítulos 2, 3 y 4). Algunas preguntas que planteamos son las siguientes:

- ¿Cuantas especies se incluyen en las áreas relevantes de diversidad definidas por cada criterio?
- ¿Cuántas especies amenazadas y por consiguiente con interés en conservación son excluidas de las áreas relevantes de diversidad definidas por cada criterio?
- ¿Qué cantidad de territorio es necesario para representar dentro de las áreas relevantes de diversidad a todas las especies y a todas las especies amenazadas?

3. Examinar mediante *gap analysis* la Red de Espacios Naturales Protegidos y la Red Natura 2000 en España (Capítulos 2 y 4).

- ¿Cuáles son las áreas relevantes de diversidad definidas por los distintos criterios que coinciden con la red de áreas de conservación?

4. Determinar en qué medida las grandes infraestructuras planeadas en España podrían afectar a la herpetofauna (Capítulo 3).

- ¿Qué porcentaje de áreas relevantes de diversidad de herpetofauna coincide con las nuevas infraestructuras?
- ¿Qué especies de anfibios y reptiles son las más afectadas?
- ¿Qué áreas deberían ser preferentemente consideradas para acciones de mitigación de impactos ambientales?

5. Desarrollar un ejemplo de planificación sistemática de la conservación en Castilla-La Mancha que incluya múltiples componentes tales como elementos especiales de conservación, especies clave y la representación de todos los hábitats importantes para el mantenimiento de la biodiversidad (Capítulo 4).

- ¿Qué tipos de hábitat están poco representados por la red de áreas de conservación?
- ¿Qué zonas son las más apropiadas para mejorar la conectividad de las áreas de conservación?
- ¿Qué áreas son necesarias para garantizar la conservación de todas las especies de vertebrados en el área de estudio?

6. Evaluar los cambios provocados por el aclaramiento o resalveo del monte mediterráneo en la diversidad de especies y diversidad funcional de la comunidad de aves a escala local (Capítulo 5).

- ¿Qué cambios provoca el resalveo en la estructura de la vegetación del monte mediterráneo?
- ¿Cuál es el efecto del resalveo en la riqueza de especies y en la composición y la abundancia de la comunidad de aves? ¿Y en los distintos gremios según su uso del hábitat y sus preferencias tróficas? ¿Y en las especies con interés de conservación a nivel europeo? ¿Y en las especies cinegéticas?
- ¿Pueden extenderse a la región Mediterránea los efectos del resalveo demostrados previamente en otras regiones forestales?
- Comprobar en condiciones naturales la hipótesis de que la complejidad estructural del hábitat causa un efecto alométrico en la masa corporal media de la comunidad de aves.

Referencias

Aebischer, N.J., Green, R.E. & Evans, A.D. 2000. From science to recovery: tour case studies of how research has been translated into conservation action in the UK. In: Ecology and Conservation of Lowland Farmland Birds (Eds. Aebischer, N.J., Evans, A.D., Grice, P.V. & Vickery, J.A.). British Ornithologists' Union, Tring, pp 43-54.

Araújo, M.B. & Williams, P.H. 2000. Selecting areas for species persistence using occurrence data. Biological Conservation 96, 331-345.

Arponen, A., Heikkinen, R.K., Thomas, C.D. & Molainen, A. 2005. The value of biodiversity in reserve selection: Representation, Species Weighting, and benefit functions. Conservation Biology 19, 2009-2014.

Atauri, J.A. & de Lucio, J.V. 2001. The role of landscape structure in species richness and distribution of birds, amphibians, reptiles and lepidopterans in Mediterranean landscapes. Landscape Ecology 16, 147-159.

Atwell, R.C., Schulte, L.A. & Palik, B.J. 2008. Songbird response to experimental retention harvesting in red pine (*Pinus resinosa*) forests. Forest Ecology and Management 255, 3621-3631.

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R.E., Jenkins, M., Jefferiss, P., Jessamy, V., Madden, J., Munro, K., Myers, N., Naeem, S., Paavola, J., Rayment, M., Rosendo, S., Roughgarden, J., Trumper, K. & Turner R.K. 2002. Economic reasons for conserving wild nature. Science 297, 950-953.

Barbier, E.B., Burgess, J.C. & Folke, C. 1994. Paradise Lost? The Ecological Economics of Biodiversity, Earthscan, London.

Beazley, K., Smandych, L., Snaith, T., Mackinnon, F., Austen-Smith, Jr.P. & Duinker, P. 2005. Biodiversity considerations in conservation system planning: map-based approach for Nova Scotia, Canada. Ecological Applications 15, 2192-2208.

Blondel, J. & Aronson, J. 1999. Biology of the wildlife of the Mediterranean region. Oxford University Press, Nex York.

Brooks, T.M., Mittermeier, R.A., da Fonseca, G.A.B., Gerlach, J., Hoffman, M., Lamoreux, J.F., Mittermeier, C.G., Pilgrim, J.D. & Rodrigues, A.S.L. 2006. Global biodiversity conservation priorities. Science 313, 58-61.

Cabeza, M., Molainen, A. & Possingham, H.P. 2004. Metapopulation dynamics and reserve network design. Pages 541-564 in I. Hanski & O. Gaggiotti (Eds.) Ecology, genetics and evolution of metapopulations. Academic Press, New York.

Callicott, J.B. 1994. Earth's Insights: A multicultural survey of ecological ethics from the Mediterranean basin to the Australian outback. University of California Press, Berkeley, CA.

Campbell, S.P., Witham, J.W. & Hunter, H.L. 2007. Long-term effects of group-selection timber harvesting on abundance of forest birds. Conservation Biology 21, 1218-1229.

Castro, I., Moreno, J.C., Humphries, C.J., & Williams, P.H. 1997. Strengthening the Natural and National Park system of Iberia to conserve vascular plants. Botanical Journal of the Linnean Society 121, 189-206.

Chapin III, F.S., Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L., Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E., Mack, M.C. & Díaz, S. 2000. Consequences of changing biodiversity. Nature 405, 234-242.

Conroy, M.J. & Noon, B.R. 1996. Mapping of species richness for conservation of biological diversity: conceptual and methodological issues. Ecological Applications 6, 763-773.

Costanza, R., d'Arge R., de Groot , R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P. & van den Belt, M. 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253-260.

Cowling, R.M., Rundel, P.W., Lamont, B.B., Arroyo, M.K. & Arianoutsou, M. 1996. Plant diversity in Mediterranean climate regions. Trends in Ecology and Evolution 11, 262-366.

Daily, G.C. 2001. Ecological forecasts. Nature 411, 245.

Daily, G.C. (Ed.) 1997. Nature's Services. Societal Dependence on Natural Ecosystems. Island Press, Washington, D.C.

Darwin, C. 1859. The Origin of Species by Means of Natural Selection (reprinted by The Modern Library, Random House, New York).

Deane-Drummond, C.E. 2004. The Ethics of Nature. Blackwell, Malden, MA.

Díaz, S., Fargione, J., Chapin III, F.S. & Tilman, D. 2006. Biodiversity loss threatens human well-being. PLoS Biol 4:e277. DOI: 10.1371/journal.pbio.0040277.

Donald, P.F., Sanderson, F.J., Burfield, I.J., Bierman, S.M., Gregory, R.D. & Waliczky, Z. 2007. International conservation policy delivers benefits for birds in Europe. Science 317, 810-813.

Drechsler, M. 2005. Probabilistic approaches to scheduling reserve selection. Biological Conservation 122, 253-262.

Faith, D.P. & Walker, P.P.A. 1996. How do indicator groups provide information about the relative biodiversity of different sets of areas?: On hotspots, complementarity and pattern-based approaches. Biodiversity Letters 3, 18-25.

Fleishman, E., Noss, R.F. & Noon, B.R. 2005. Utility and limitations of species richness metrics for conservation planning. Ecological Indicators 6, 543-553.

González Molina, J.M. 2005. Introducción a la selvicultura general. Universidad de León, Ponferrada.

Grenyer, R., Orme, C.D.L., Jackson, S.F., Thomas, G.H., Davies, R.G., Davies, T.J., Jones, K.E., Olson, V.A., Ridgely, R.S., Rasmussen, P.C., Tzung-Su Ding, Bennett, P.M., Blackburn, T.M., Gaston, K.J., Gittleman, J.L. & Owens, I.P.F. 2006. Global distribution and conservation of rare and threatened vertebrates. Nature 444, 93-96.

Halladay, P. & Gilmour, D.A. (Eds.) 1995. Conserving Biodiversity Outside Protected Areas. The role of traditional agro-ecosystems. IUCN, Gland, Switzerland, and Cambridge, UK.

Harcourt, A.H. 2000. Coincidence and mismatch of biodiversity hotspots: A global survey for the order, primates. Biological Conservation 93, 163-175.

Hargrove, E.C. 1989. Foundations of Environmental Ethics. Prentice-Hall, Englewood Cliffs, NJ.

Harris, G.M., Jenkins, C.N. & Pimm, S.L. 2005. Refining biodiversity conservation priorities. Conservation Biology 19, 1957-1968.

Heywood, V.H. 1995. Global Biodiversity Assessment. Cambridge University Press, Cambridge.

Hoeskstra, J.M., Boucher, T.M., Ricketts, T.H. & Roberts, C. 2005. Confronting a biome crisis: global disparities of habitat loss and protection. Ecology Letters 8, 23-29.

Houghton, J.T., Callander, B.A. & Varney, S.K. 1992. Climate Change 1992. The Supplementary Report to the IPCC Scientific Assessment. Cambridge Univ. Press, Cambridge, U.K.

Howard, P.C. 1991. Nature Conservation in Uganda's Tropical Forest Reserves. IUCN, Gland, Switzerland, and Cambridge, UK.

IUCN. 1980. World Conservation Strategy: living resource conservation for sustainable development, Gland: IUCN- UNEP-WWF.

James, A.N., Gaston, K.J. & Balmford, A. 1999. Balancing the Earth's accounts. Nature 401, 323-324.

Jetz, W., Rahbek, C. & Colwell, R.K. 2004. The coincidence of rarity and richness and the potential signature of history in centres of endemism. Ecology Letters 7, 1180-1191.

Jiguet, F. & Julliard, R. 2006. Inferences from common species communities for selecting conservation areas. Biodiversity and Conservation 15, 799-815.

Kleijn, D. & Sutherland, W.J. 2003. How effective are European agri-environment schemes in conserving and promoting biodiversity? Journal of Applied Ecology 40, 947-969.

Kleijn, D., Baquero, R.A., Clough, Y., Díaz, M., De Esteban, J., Fernández, F., Gabriel, D., Herzog, F., Holzschuh, A., Jöhl, R., Knop, E., Kruess, A., Marshall, E.J.P., Steffan-Dewenter, I., Tscharntke, T., Verhulst, J., West, T.M. & Yela, J.L. 2006. Mixed biodiversity benefits of agri-environment schemes in five European countries. Ecology Letters 9, 243-254.

Lamoreux, J.F., Morrison, J.C., Ricketts, T.H., Olson, D.M., Dinerstein, E., McKnight, M.W. & Shugart, H.H. 2006. Global test of biodiversity concordance and the importance of endemism. Nature 440, 212-214.

Larsen, F.W., Bladt, J. & Rahbek, C. 2007. Improving the performance of indicator groups for the identification of important areas for species conservation. Conservation Biology 21, 731-740.

Lawton, J.H. & May, R.M. 1995. Extinction Rates. Oxford University Press, Oxford.

Lovelock, J. 1979. Gaia: A New Look at Life on Earth.

Loreau, M., Naeem, S. & Inchausti, P. (Eds). 2002. Biodiversity and ecosystem functioning: synthesis and perspectives. Oxford University Press, Oxford.

Lubchenco, J., Olson, A.M., Brubaker, L.B., Carpenter, S.R., Holland, M.M., Hubbell, S.P., Levin, S.A., MacMahon, J.A., Matson, P.A., Melillo, J.M., Mooney, H.A., Peterson, C.H., Pulliam, H.R., Real, L.A., Regal, P.J. & Risser, P.G. 1991. The sustainable biosphere initiative: an ecological research agenda. Ecology 72, 371-412.

Mace, G.M., Balmford, A., Boitani, L., Cowlishaw, G., Dobson, A.P., Faith, D.P., Gaston, K.J., Humphries, C.J., Vane-Wright, R.I., Williams, P.H., Lawton, J.H., Margules, C.R., May, R.M., Nicholls, A.O., Possingham, H.P., Rahbek, C., van Jaarsveld, A.S. 2000. It's time to work together and stop duplicating conservation efforts… Nature 405, 393.

Magurran, A.E. 1989. Diversidad Ecológica y su Medición. Vedrá, Barcelona.

Margalef, R. 1974. Ecología. Omega, Barcelona.

McCann, K.S. 2000. The diversity-stability debate. Nature 405, 228-233.

McNeely, J.A, Miller, K.R., Reid, W.V., Mittermeier, R.A. & Werner, T.B. 1990. Conserving the World's Biological Diversity, IUCN, World Resources Institute, CI, WWF-US, the World Bank, Gland, Switzerland and Washington, D.C.

Millennium Ecosystem Assessment. 2005. Ecosystems and human well-being: Biodiversity synthesis. Washington, D.C.: World Resources Institute. 86p.

Milner-Gulland, E.J., Kreuzberg-Mukhina, E., Grebot, B., Ling, S., Bykova, E., Abdusalamov, I., Bekenov, A., Gardenfors, U., Hilton-Taylor, C., Salnikov, V. & Stogova, L. 2006. Application of UICN red listing criteria at the regional and national levels: A case study from central Asia. Biodiversity and Conservation 15, 1873-1886.

Murdoch, W., Polasky, S., Wilson, K.A., Possingham, H.P., Kareiva, P., & Shaw, R. 2007. Maximizing return on investment in conservation. Biological Conservation 139, 375-388.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A. & Kent, J. 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853-858.

Olson, D.M. & Dinerstein, E. 2002. The global 200: priority ecoregions for global conservation. Annals of the Missouri Botanical Garden 89, 199-224.

Orme, C.D.L., Davies, R.G., Burgess, M., Eigenbrod, F., Pickup, N., Olson, V.A., Webster, A.J., Tzung-Su Ding, Rasmussen, P.C., Ridgely, R,S., Stattersfield, A.J., Bennett, P.M., Blackburn, T.M., Gaston, K.J. & Owens, I.P.F. 2005. Global hotspots of species richness are not congruent with endemism or threat. Nature 436, 1016-1019.

Peach, W.J., Lovett, L.J., Wotton, S.R. & Jeffs, C. 2001. Countryside stewardship delivers cirl buntings (Emberiza cirlus) in Devon, UK. Biological Conservation 101, 361-373.

Pimentel, D., Stachow, U., Takacs, D.A., Brubaker H.W., Dumas, A.R., Meaney, J.J., O'Neil, J.A.S., Onsi, D.E. & Corzilius, D.B. 1992. Conserving biological diversity in agricultural/forestry systems. BioScience 42, 354-362.

Pimentel, D., Wilson, C., McCullum, C., Huang, R., Dwen, P., Flack, J., Tran, Q., Saltman, T. & Cliff, B. 1997. Economic and environmental benefits of biodiversity. BioScience 47, 747-757.

Pimm, S.L. 1991. The Balance of Nature? University of Chicago Press, Chicago.

Pimm, S.L., Russell, G.J., Gittleman, J.L. & Brooks, T.M. 1995. The future of biodiversity. Science 269, 347-350.

Prendergast, J.R., Quinn, R.M., Lawton, J.H., Eversham, B.C. & Gibbons, D.W. 1993. Rare species, the coincidence of diversity hotspots and conservation strategies. Nature 365, 335-337.

Rabinowitz, D. 1981. Seven forms of rarity. Pages 205-217 in H. Synge, editor. The biological aspects of rare plant conservation. John Wiley, Chichester, UK.

Reid, W.V. 1998. Biodiversity hotspots. Trends in Ecology and Evolution 13, 275-280.

Rey Benayas, J.M., Scheiner, S.M., García Sánchez-Colomer, M. & Levassor, C. 1999. Commonness and rarity: theory and application of a new model to Mediterranean montane grasslands. Conservation Ecology [online] 3(1):5. Available at www.consecol.org/vol3/iss1/art5.

Rodrigues, A.S., Andelman, S.J., Bakarr, M.I., Boitani, L., Brooks, T.M., Cowling, R.M., Fishpool, L.D., Da Fonseca, G.A., Gaston, K.J., Hoffmann, M., Long, J.S., Marquet, P.A., Pilgrim, J.D., Pressey, R.L., Schipper, J., Sechrest, W., Stuart, S.N., Underhill, L.G., Waller, R.W., Watts, M.E. & Yan, X. 2004. Effectiveness of the global protected area network in representing species diversity. Nature 428, 640-643.

Rundel, P.W. 1998. Landscape Disturbance in Mediterranean-Type Ecosystems: An Overview. In Rundel, P.W., Montenegro, G. & Jaksic, F.M., (Eds.) Landscape Disturbance and Biodiversity in Mediterranean-Type Ecosystems. Springer-Verlag Berlin. pp 3-22.

Sala, O.E., Chapin III, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., LeRoy Poff, N., Sykes M.T., Walker, B.H., Walker, M. & Wall, D.H. 2000. Global biodiversity scenarios for the year 2100. Science 287, 1770-1774.

Salomon, A.K., Ruesink, J.L. & DeWreede, R.E. 2006. Population viability, ecological processes and biodiversity: Valuing sites for reserve selection. Biological Conservation 128, 79-92.

San Miguel (Coord.). 2006. Manual para la gestión del hábitat del lince ibérico (*Lynx pardinus* Temminck) y de su presa principal el conejo de monte (*Oryctolagus cuniculus* L.) Fundación CBD-Habitat, Madrid.

Sarkar, S. 2005. Biodiversity and environmental philosophy. An introduction. Cambridge University Press, Cambridge.

Sarkar, S., Aggarwal, A., Garson, J., Margules, C.R. & Zeidler, J. 2002. Place prioritization for biodiversity content. Journal of Biosciences 27, 339-346.

Sekercioglu, C.H. 2002. Effects of forestry practices on vegetation structure and bird community of Kibale National Park, Uganda. Biological Conservation 107, 229-240.

Schulze, E.D. & Mooney, H.A. 1993. Biodiversity and Ecosystem Function. Springer-Verlag, Berlin.

Sergio, F., Newton, I. & Marchesi, L. 2005. Top predators and biodiversity. Nature 436, 192.

Serrada, R., Montero, G. & Reque, J.A. 2008. Compendio de selvicultura aplicada en España. Instituto Nacional de Investigación y Tecnología Agraria y Alimentaria, Madrid.

Statterfield, A.J., Crosby, M.J., Long, A.J. & Wege, D.C. 1998. Endemic Bird Areas of the World: Priorities for Biodiversity Conservation. Cambridge, UK: Birdlife International. pp 846.

Strange, N., Thorsen, B.J. & Bladt, J. 2006. Optimal reserve selection in a dynamic world. Biological Conservation 131, 33-41.

Thompson, I.D., Baker, J.A. & Ter-Mikaelian, M. 2003. A review of the long-term effects of post-harvest silviculture on vertebrate wildlife, and predictive models, with an emphasis on boreal forests in Ontario, Canada. Forest Ecology and Management 177, 441-469.

Tilman, D. 1999. The ecological consequences of change in biodiversity: A search for general principles. Ecology 80, 1455-1474.

Tilman, D. 2000. Causes, consequences and ethics of biodiversity. Nature 405, 208-211.

Underwood, E.C., Shaw, M.R., Wilson, K.A. Kareiva, P., Klausmeyer, K.R., McBride, M.F., Bode, M., Morrison, S.A., Hoekstra, J.M. & Possingham, H.P. 2008. Protecting biodiversity when money matters: maximizing return on investiment. PLoS ONE 3, e1515. doi:10.1371/journal.pone.0001515.

UNCED. 1992. United Nations Conference on Environment and Development. Río de Janeiro. Available at www.biodiv.org.

UNEP. 2002. "Report on the Sixth Meeting of the Conference of the Parties to the Convention on Biological Diversity (UNEP/CBD/COP/6/20/Part 2) Strategic Plan Decision VI/26"; available at www.biodiv.org/doc/meetings/cop/cop-06/official/cop-06-20-part2-en.pdf.

UNEP. 2004. "Report on the Seventh Meeting of the Conference of the Parties to the Convention on Biological Diversity (UNEP/CBD/COP/7/21/Part 2) Strategic Plan Decision VII/30"; available at www.biodiv.org/decisions/dfault.aspx

Vickery, J.A., Bradbury, R.B., Henderson, I.G., Eaton, M.A. & Grice, P.V. 2004. The role of agri-environment schemes and farm management practices in reversing the decline of farmland birds in England. Biological Conservation 119, 19-39.

Virolainen, K.M., Nättinen, K., Suhonen, J. & Kuitunen, M. 2001. Selecting herb-rich forest networks to protect different measures of biodiversity. Ecological Applications 11, 411-420.

Vitousek, P.M., Mooney, H.A., Lubchenco, J. & Melillo, J.M. 1997. Human domination of earth's ecosystems. Science 277, 494-499.

Wallace, A.R. 1876. The Geographical Distribution of Animals. With a study of the relations of living and extinct faunas as elucidating the past changes of the earth's surface. Macmillan & Co. London.

Williams, J., ReVelle, C.S. & Levin, A. 2004. Using mathematical optimization models to design nature reserves. Frontiers in Ecology and the Environment 2, 98-105.

Williams, P., Gibbons, D., Margules, C., Rebelo, A., Humphries, C. & Pressey, R. 1996. A comparation of richness hotspots, rarity hotspots, and complementary areas for conserving diversity using British birds. Conservation Biology 10, 155-174.

Williams, P.H. & Araújo, M.B. 2000. Using probabilities of persistence too identify important areas for biodiversity conservation. Proceedings of the Royal Society of London Biological Sciences 267, 1959-1966.

Wilson, E.O. (Ed.) 1988. Biodiversity. Washington, DC: National Academy of Sciences/Smithsonian Institution.

Wilson, K.A., McBride, M.F., Bode, M. & Possingham, H.P. 2006. Prioritising global conservation efforts. Nature 440, 337-340.

Xu, H.G., Wu, J., Liu, Y., Ding, H., Zhang, M., Wu, Y., Xi, Q. & Wang, L.L. 2008. Biodiversity congruence and conservation strategies: A national test. Bioscience 58, 632-639.

Capítulo 2

"La vida de todos los seres, sean humanos, animales o de otra clase es preciosa y todos tienen el mismo derecho a la felicidad. Los pájaros, los animales salvajes... Todos lo que pueblan nuestro planeta nos acompañan. Forman parte de nuestro mundo, lo compartimos con ellos"

XIV Dalai Lama

Jeroglífico egipcio con representaciones de animales indicando el nombre del faraón. Siglo VI a.c. Extraído de Sánchez Torrente, P., González Encinas, A.L. 1998. Grandes Civilizaciones de la Antiguedad. Egipto.

Capítulo 2

Identificación de áreas relevantes de diversidad de vertebrados para el fortalecimiento de la conservación

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

José M. Rey Benayas, Enrique de la Montaña (2003). Identifying areas of high-value vertebrate diversity for strengthening conservation. Biological Conservation 114, 357-370.

Resumen

La identificación de áreas con relevantes características de biodiversidad es útil para decidir prioridades que mejoren el diseño de espacios naturales protegidos y para optimizar los recursos invertidos en conservación. Este estudio aporta herramientas claves a los gestores para señalar qué áreas son merecedoras de conservación en España. Estudiamos cuatro taxones -anfibios, reptiles, aves y mamíferos- en una malla con celdas de 50 x 50 km (*n*= 259 celdas). Usamos cinco criterios para identificar las áreas relevantes de diversidad: riqueza de especies, rareza, vulnerabilidad, un índice combinado de biodiversidad, y un Índice Estandarizado de Biodiversidad que evalúa los cuatro taxones juntos. Se proponen dos nuevos índices: el índice combinado de biodiversidad y el Índice Estandarizado de Biodiversidad. Se definieron las áreas relevantes de diversidad como aquellas celdas situadas en el 15% superior de los datos ordenados según los diferentes criterios. La congruencia entre las áreas relevantes de diversidad de los taxones analizados por parejas fue de moderada a baja, y de un 38.5% de media para las áreas relevantes de diversidad basadas en el índice combinado de biodiversidad. Los resultados basados en la proporción media de las especies amenazadas excluidas de las áreas relevantes de diversidad siguió este orden: índice combinado de biodiversidad = rareza (0.3%) > vulnerabilidad (9.9%) > riqueza de especies (13.8%). Las áreas relevantes de diversidad identificadas según el Índice Estandarizado de Biodiversidad incluyeron todas las especies de anfibios y mamíferos, todas las especies de reptiles excepto una (catalogada como rara) y todas las especies de aves excepto seis (tres de las cuales estaban catalogadas como amenazadas). Alrededor del 70% de las áreas relevantes de diversidad identificadas según el Índice Estandarizado de Biodiversidad incluyeron espacios naturales protegidos. Sin embargo, la superficie media de estas áreas protegidas es de 2746 km^2, por lo que ocupan una pequeña fracción de las áreas relevantes de diversidad, y no hay garantías de que las especies halladas en un área relevante de diversidad estén presentes en la fracción protegida. Por lo tanto, instamos a los gestores de los espacios naturales protegidos a realizar inventarios de diversidad. También recomendamos que sean establecidos espacios naturales protegidos adicionales que incluyan ese 30% de áreas relevantes de diversidad que actualmente no están protegidas. Mostramos un primer paso para la planificación de la conservación de la biodiversidad en la región estudiada, y discutimos la utilidad de los mapas de áreas relevantes de diversidad para la conservación, restauración ecológica, y mitigación y evaluación de impactos ambientales.

Palabras clave: Espacios naturales protegidos; Especies amenazadas; Índice estandarizado de biodiversidad; Rareza; Riqueza de especies; Vertebrados; Vulnerabilidad

Identifying areas of high-value vertebrate diversity for strengthening conservation

José M. Rey-Benayas, Enrique de la Montaña

Departamento de Ecología, Universidad de Alcalá. Alcalá de Henares, 28871 Madrid, Spain.

Abstract

Identifying areas with relevant features of biodiversity is useful to rank priorities for strengthening the design of well-sited natural protected areas and to optimize resource investment in conservation. This study provides decision makers critical tools for highlighting pieces of land worthy of conservation in Spain. We studied four taxa -amphibians, reptiles, nesting birds and mammals- in a 50 x 50 km grid (n = 259 cells). We used five criteria for identifying areas of high-value diversity: species richness, rarity, vulnerability, a combined index of biodiversity, and a Standardised Biodiversity Index that measured all four taxa together. As far as we know, the combined index of biodiversity and the Standardised Biodiversity Index are original. Areas of high-value diversity were defined as those cells within the 15% top segment of ranked values for the different criteria. Congruence of areas of high-value diversity for taxa pairs was moderate to low, and averaged 38.5% for areas of high-value diversity based on the combined index of biodiversity. The performance based on the average proportion of threatened species excluded from areas of high-value diversity followed the rank combined index of biodiversity = rarity (0.3%) > vulnerability (9.9%) > species richness (13.8%). The areas of high-value diversity identified according to the Standardised Biodiversity Index included all amphibian and mammal species, all but 1 reptile species (categorized as rare) and all but 6 bird species (3 of which were categorized as threatened). About 70% of the areas of high-value diversity identified based on the Standardised Biodiversity Index included natural protected areas. However, they average only 274.6 km^2, thus occupying a small fraction of the areas of high-value diversity, and there is no guarantee that the species found in an area of high-value diversity site will be present in its protected fraction. Consequently, we urge managers of natural protected areas to conduct diversity surveys. We also urge that additional natural protected areas be established to include the gap of 30% of areas of high-value diversity not currently protected. We took an step for biodiversity conservation planning in the studied region, and discuss the usefulness of maps of areas of high-value diversity for conservation, ecological restoration, and environmental impact assessment and mitigation.

Keywords: Gap analysis; Natural protected areas; Rarity; Species richness; Standardized biodiversity index; Threatened species; Vertebrates; Vulnerability

Introduction

The scientific community has witnessed a considerable debate about biodiversity conservation during the last decade. The debate has been fuelled by studies of the increasing rates of biodiversity loss (Heywood, 1995; Costanza *et al*., 1997; Pimentel *et al*., 1997; Ricketts, 1999; Terborgh, 1999; Tilman, 1999; Bininda-Emonds et *al*., 2000; Cincotta *et al*., 2000; Myers *et al*., 2000; Pimm & Raven, 2000). Biodiversity conservation usually requires decision making that needs input by both conservation biologists and ecologists. Reserves alone are not adequate for nature conservation but they are the cornerstone on which regional strategies can be built (Margules & Pressey, 2000), and it is necessary to set prior targets (Ceballos *et al*., 1998; Myers *et al*., 2000). We address this problem of setting prior targets by identifying areas of high-value diversity of vertebrate species in Spain, evaluating the effectiveness of various criteria used to define them, and examining their geographical coincidence with the existing network of natural protected areas. Thus, we hope to provide decision makers with critical tools for identifying tracts of land particularly worthy of conservation.

Detecting areas of high-value diversity is useful, in our view, as a tool for four major tasks related to nature conservation. These are: 1) creating networks of natural protected areas, 2) conservation out from natural protected areas favored by a wise natural resource management, 3) ecological restoration as an action to preserve threatened species and communities, and 4) environmental impact assessment and mitigation. This paper focuses on the first topic. Establishing natural protected areas is one of the most useful tools for preserving large pools of biodiversity. Unfortunately, throughout most of the world there are neither the resources nor the time to carry out detailed inventories for most *taxa* before designating protected areas (Groombridge, 1992; Raven & Wilson, 1992; Prendergast *et al*., 1993). Besides, this is often expensive because land needs to be purchased and managed for species conservation (Mittermeier, 1998; James *et al*., 1999). Therefore, it is important to rank priorities of sites with relevant features of biodiversity to optimize resource investment in conservation. These relevant features of biodiversity are most frequently based on criteria such as species richness, rarity (particularly endemic *taxa*), taxonomic uniqueness, threatened species and indicator *taxa* (Usher, 1986; Williams *et al*., 1991; Prendergast *et al*., 1993; Faith & Walker, 1996; Castro *et al*., 1997; Reid, 1998; Rey Benayas *et al*., 1999; Virolainen *et al*., 2001).

To accomplish our objectives it is necessary, first, to identify what areas are relevant for their biodiversity features. We used four *taxa* of vertebrate species: amphibians, reptiles, breeding birds and mammals. These are well studied *taxa* because they are conspicuous. Birds are also "popular", whereas amphibians and reptiles figure less prominently in site selection. Many of these species have been used as bioindicators. Particularly, mammals such as *Ursus arctos*, *Lutra lutra* and *Lynx pardinus* are outstanding indicators of habitat quality in the study area. We used five criteria for identifying areas of high-value diversity: species richness, rarity, vulnerability, a combined index of the three former, and a Standardized Biodiversity Index that measured all four *taxa* together. Our analyses are illustrative, not exhaustive. A similar approach can be taken to identify areas of high-value diversity elsewhere, using either different species groups or criteria.

Site selection would be easier if there was overlap of areas of high-value diversity of different *taxa* and areas of high-value diversity identified according to different criteria. We evaluated the effectiveness of the various criteria used to identify areas of high-value diversity by looking at the species lists that were included and excluded. Finally, we overlaid the produced maps of areas of high-value diversity with the existing network of natural protected areas to examine coincidences and highlight gaps. Our intention is to provide useful information for diversity conservation to decision makers (politicians, technicians, land managers, managers of natural protected areas, etc.).

The study area and its existing network of natural protected areas

The Iberian Peninsula and the Balearic Islands include a variety of biomes, relief, climates, and soil types despite their relatively small area (585644 km^2 in total). Two major climatic zones, Mediterranean and Atlantic, impinge on Iberia (Font Tullot, 1983). The Mediterranean climate is characteristic of most of Iberia and the Balearic Islands, whereas the Atlantic climate is found in a band ca. 100 km wide along the western and northern coasts, and it also influences the Pyrenean Mountains. The current driest and warmest areas served as a *refugium* during Pleistocene glaciations. In contrast, the northern transition zone is substantially younger, having emerged only after the glacial retreat.

Spain is one of the most diverse countries in the European Union. The Canary Islands will not be considered in this study because they represent Macaronesic biotas completely different to the rest of the country. The Spanish network of natural protected areas currently comprises 33304 km^2. About 86.2% of this land (28721 km^2) belongs to 12 National Parks, 91 Natural Parks -including 1 Biosphere Reserve- and 11 Regional Parks (Fig. 2.1 and Appendix 2.A) (Gómez-Limón *et al*., 2000). Their mean area in continental Spain and the Balearic Islands is 274.6 ± 419.1 km^2, and range between 0.45 and 2143.4 km^2. Four National Parks and 11 Natural Parks are located in the Canary Islands. By law, national, natural and regional parks must implement a management plan of natural resources, and hence are the most robust figures of natural protected areas.

Materials and Methods

Analytical units and data sources

Our analyses used cells defined by Universal Transverse Mercatore (UTM) coordinates as analytical units (Fig. 2.1). The modal cell size was 2500 km^2. This is the smallest size common to all *taxa* considered in this study. The cells adjacent to borders between different UTM coordinate zones are slightly different in size. We examined the presence and absence of amphibian, reptile, breeding bird, and mammal species in a total of 259 cells. The cell-by-species matrices were built using species distribution atlases (Gasc *et al*., 1997; Hagemeijer and Blair, 1997; Mitchell-Jones *et al*., 1999). We counted 27, 42, 269, and 89 amphibian, reptile, breeding bird, and mammal species, respectively. Information on species presence for breeding birds and mammals was not available for 2 and 6 cells, respectively.

Criteria for identifying areas of high-value diversity

We used four criteria to identify areas of high-value diversity for a single *taxa*: species richness, rarity, vulnerability, and a combined index of biodiversity. There are many forms of rarity, that respond to different combinations of geographical range, local abundance, habitat specificity and habitat occupancy (Rabinowitz, 1981; Rey Benayas *et al.*, 1999). In this study, rarity of species *i* was defined by its geographical range measured as the inverse of the number of cells where it was present ($1/n_i$). Currently, there are not official criteria in Spain classifying species into rarity categories according to their geographical ranges (Perring & Farrel, 1983; Cameron, 1998). For a cell *r*, the rarity index was $\sum_{i=1}^{S}(1/n_{ri})/s_r$, where s_r was the number of species found in the cell.

Vulnerability of a species was quantified using the categories of the *Red Book of Spanish Vertebrates* (Blanco & González, 1992). The species categories that were considered are the following: endangered, vulnerable, rare, undetermined, insufficiently known, introduced, and non-threatened. These categories were previously defined by the International Union for Conservation of Nature (IUCN, 1988). A complete list of the species studied here and their vulnerability category and number of cells where they were found are available from the autors upon request. Currently, these categories are

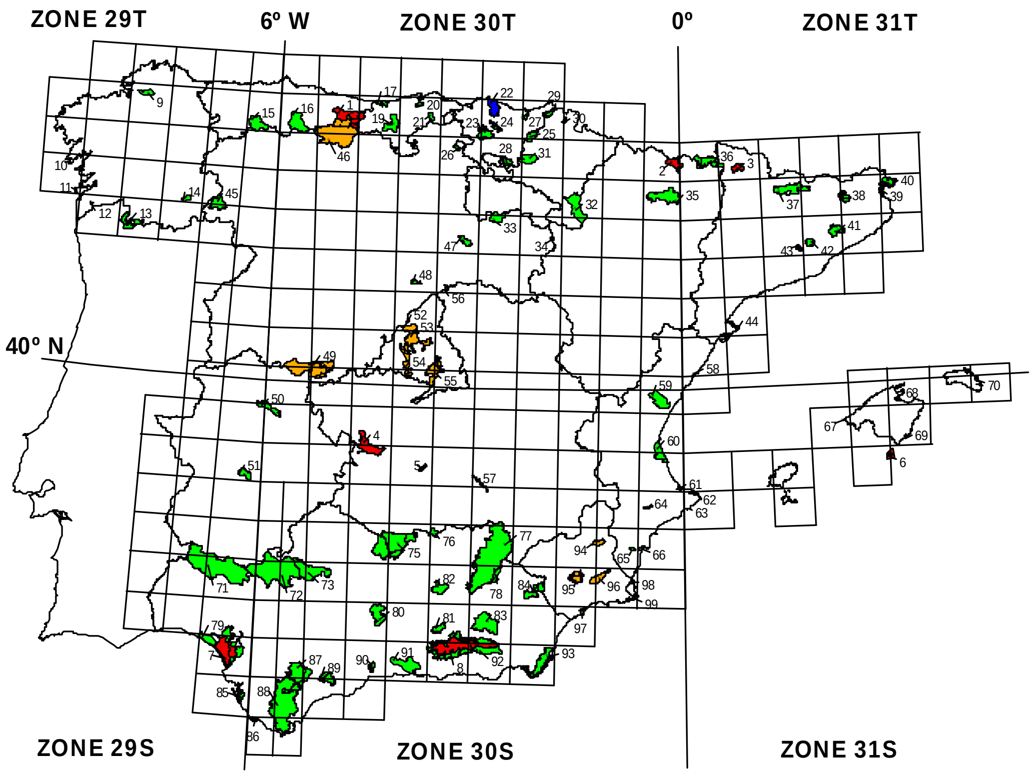


Figure 2.1. Map of continental Spain and the Balearic Islands (area is 497508 km^2). It illustrates the grid used for the analysis of areas of high-value diversity -based on UTM coordinates-, the political distribution of the territory (autonomous regions), and the location of national, natural and regional parks (red, green, and orange spots, respectively; the blue spot is a Biosphere Reserve). Source of the location of the natural protected areas was Gómez-Limón *et al.* (2000). A complete list of the natural protected areas considered in this study can be found in Appendix 2.A.

being revised, but no publications have been released yet. We assigned to every category a score related to its degree of vulnerability, ranging from 5 for endangered species to 1 for non-threatened and introduced species. The intermediate categories were assigned 4 (vulnerable and undetermined), 3 (rare) and 2 (insufficiently known). Since the undetermined species are those that are known to belong to the endangered, vulnerable, or rare categories, we decided to assign them the score = 4 because it is the average score of these categories. We acknowledge the subjectivity of these scores; they just represent a rank and have a relative value, and any other choice would have been subjective as well. For a cell, the vulnerability index was $\sum_{i=1}^{s} V_{ri}/s_r$, where V_{ri} was the vulnerability score of the species present in the cell. As vulnerability is actually a surrogate concept of rarity plus rates of habitat loss and other threats, we expect a positive correlation between these two indices. For every cell and taxonomic group, we defined a combined index of species richness, rarity, and vulnerability: $\sum_{i=1}^{s}(1/n_{ri})V_{ri}$. In this index, species richness is implicit in $\sum_{i=1}^{s}$. As far as we know, this index is original. Since the combined index of biodiversity is a function of three other indices, we expect in general a positive correlation between this combined index and richness, rarity, and vulnerability.

We also used a Standardised Biodiversity Index that measured species richness, rarity and vulnerability of all four *taxa* together in every cell. This index is original as well. We standardised by dividing the combined index of biodiversity of each taxonomic group in every cell by its mean across all cells, and then summed the four standardised combined indices. The Standardised Biodiversity Index formula is:

$$\sum_{j=1}^{4} 1/m_j \sum_{i=1}^{jS} (1/n_{ji})V_{ji}$$

where m_j refers to the mean combined index of biodiversity of the taxonomic group j across cells. Next, all indices for the various *taxa* and the Standardised Biodiversity Index were ranked. We considered areas of high-value diversity those cells within the upper 15% (39 cells) of ranked values for the various criteria. This percentage was chosen because the commonly used 10 or 12% is considered insufficient to achieve conservation goals (Soulé & Sanjayan 1998; Margules & Pressey 2000). In spite of that, we acknowledge the arbitrarity of this threshold; any other choice would have been arbitrary as well. Statistics of all criteria used to identify areas of high-value diversity in the study area are reported in Table 2.1. Our identification of these

Table 2.1. Statistics of all criteria used to identify areas of high-value diversity in the study area.

	Amphibians (*n*=259)			Reptiles (*n*=259)			Breeding birds (*n*=257)			Mammals (*n*=253)		
	Mean	S.D.	Range	Mean	S.D.	Range	Mean	S.D.	Range	Mean	S.D.	Range
Richness	8.91	3.79	0-16	14.69	5.16	2-25	112.26	32.85	13-175	28.12	13.77	1-65
Rarity	0.013	0.026	0-0.372	0.012	0.011	0.0047-0.083	0.0097	0.0066	0.0047-0.057	0.011	0.0037	0.0058-0.031
Vulnerability	1.08	0.283	0-3	1.153	0.269	1-2.8	1.439	0.145	1.1-2	1.734	0.405	1-3.33
Combined Index	0.153	0.363	0-5.337	0.282	0.359	0.0095-2.263	1.965	1.836	0.11-23.1	0.699	0.602	0.0058-4.19

For the Standardised Biodiversity Index of all taxa, statistics are: mean = 4.04, S.D.= 3.60, range = 0.203-36.746 (*n* = 253).

areas is not affected by the different species richness of the *taxa* analysed in this study because we ultimately used ranks.

Data analysis

We examined the relationships between the four criteria within *taxa* and across *taxa* by means of correlation analysis using Bonferroni corrections for multiple comparisons. These correlations will hint the macroecological trends of diversity in the area. The congruence between areas of high-value diversity for different *taxa* was examined by means of contingency tables. To evaluate the effectiveness of the various criteria used to identify areas of high-value diversity, we examined the proportions of species included in them and threatened species that were excluded from them. For our purposes, "threatened species" were considered to be those belonging to the endangered, vulnerable, rare, and undetermined categories of IUCN (1988). Finally, we looked at the coincidence between areas of high-value diversity and national, natural and regional parks using contingency tables to highlight possible gaps.

Results

Correlations between criteria used to identify areas of high-value diversity

Within *taxa*, after applying corrections for multiple comparisons, the pair-wise correlation coefficients between the different criteria used to identify areas of high-value diversity were usually significant at $P<0.05$ (Table 2.2). The exceptions were species richness and vulnerability, which were not correlated for any taxonomic group, and species richness and rarity,

Table 2.2. Correlation coefficients between criteria used to identify areas of high-value diversity within *taxa*.

	Amphibians (*n*=259)			**Reptiles** (*n*=259)			**Breeding birds** (*n*=257)			**Mammals** (*n*=253)		
	Richn.	Rarity	Vulner.	Richn.	Rarity	Vulner.	Richn.	Rarity	Vulner.	Richn.	Rarity	Vulner.
Amphibians												
Rarity	-0.106											
Vulnerability	-0.088	**0.736**										
Combined Index	0.117	**0.917**	**0.564**									
Reptiles												
Rarity				-0.163								
Vulnerability				-0.052	**0.778**							
Combined Index				**0.227**	**0.744**	**0.718**						
Breeding birds												
Rarity							-0.183					
Vulnerability							-0.037	**0.296**				
Combined Index							**0.367**	**0.615**	**0.324**			
Mammals												
Rarity										**0.639**		
Vulnerability										-0.027	**0.243**	
Combined Index										**0.247**	**0.841**	0.197

Coefficients in bold are significant at $P<0.05$ after applying Bonferroni´s corrections for multiple comparisons. Underlined correlation coefficients are significant at $P<0.05$ if corrections for multiple comparisons were not applied.

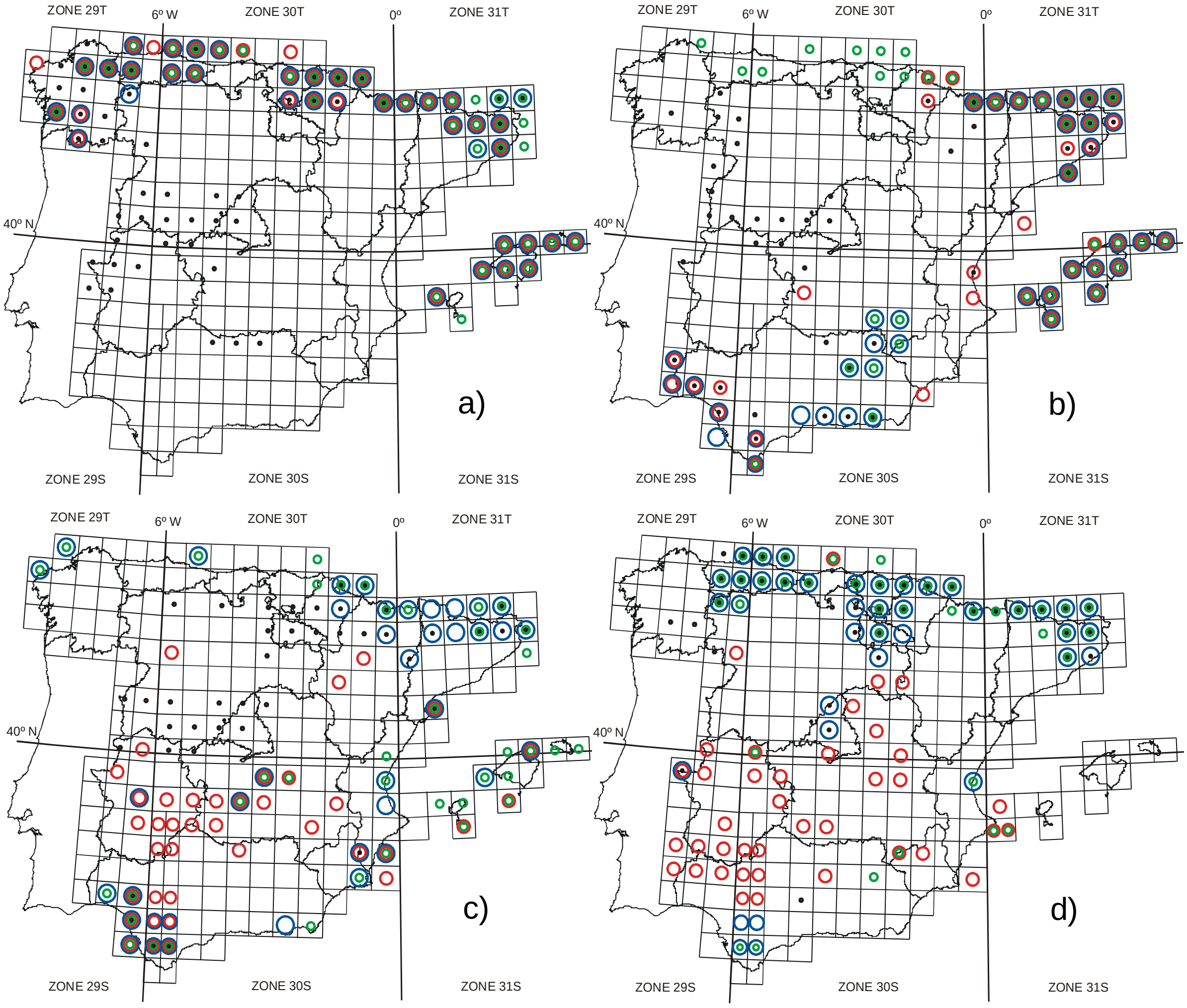


Figure 2.2. Maps of areas of high-value diversity (AHVD) identifyed according to different criteria for a) amphibians, b) reptiles, c) breeding birds, and d) mammals. Symbols are the following: black solid dots are species richness AHVD, green rings are rarity AHVD, red rings are vulnerability AHVD, and blue rings are AHVD according to the combined index of biodiversity.

which were not correlated for amphibian, reptile, and breeding bird species. That is, the richest communities did not necessarily include the rarest and most vulnerable species. As expected, rarity and vulnerability were significantly correlated, but the correlation for breeding birds and mammals were low (Table 2.2). The combined index of biodiversity was significantly correlated with the other three criteria except in the cases of species richness of amphibians and vulnerability of mammals. Thus, in general, cells with high values of the combined index of biodiversity also included rich communities with rare and vulnerable species. This indicates that a cell that has been identified as an area of high-value diversity based on one criterion (e.g., species richness), may not be an area of high-value diversity if based on a different criterion (e.g. rarity) (Fig. 2.2a-d).

Table 2.3. Correlation coefficients for criteria used to identify areas of high-value diversity between *taxa*. Presentation as in Table 2.2.

	Amphibians (*n*=259)				**Reptiles** (*n*=259)				**Breeding birds** (*n*=257)			
	Richn.	Rarity	Vulner.	Combined index	Richn.	Rarity	Vulner.	Combined index	Richn.	Rarity	Vulner.	Combined index
Reptiles												
Richness	**0.686**											
Rarity		**0.429**										
Vulnerability			**0.398**									
Combined Index				**0.229**								
Breeding birds												
Richness	**0.487**				**0.571**							
Rarity		**0.262**				**0.53**						
Vulnerability			<u>-0.188</u>				<u>0.156</u>					
Combined Index				0.067				**0.261**				
Mammals												
Richness	**0.575**				**0.468**				**0.501**			
Rarity		0.086				<u>0.191</u>				0.087		
Vulnerability			<u>-0.149</u>				<u>-0.196</u>				0.016	
Combined Index				<u>0.173</u>				**0.26**				**0.217**

Table 2.4. Percentage congruence between areas of high-value diversity of pairs of *taxa* identified according to different criteria.

	Amphibians				**Reptiles**				**Breeding birds**			
	Richn.	Rarity	Vulner.	Combined index	Richn.	Rarity	Vulner.	Combined index	Richn.	Rarity	Vulner.	Combined index
Reptiles												
Richness	47.7****											
Rarity		61.5****										
Vulnerability			46.2****									
Combined Index				41.0****								
Breeding birds												
Richness	43.9****				43.9****							
Rarity		48.7****				48.7****						
Vulnerability			2.6				17.9					
Combined Index				35.9**				41.0****				
Mammals												
Richness	43.6****				30.7				33.3*			
Rarity		53.8****				43.6****				30.8*		
Vulnerability			2.6				2.6				25.6	
Combined Index				51.3****				23.1				38.5**

The statistical significance of the associations according to χ^2 tests is indicated (*P<0.05, **P<0.01, ****P<0.0001).

Among *taxa*, the pair-wise correlations between the same criteria were very different (Table 2.3). After applying Bonferroni´s correction, the species richnesses of *taxa* were always significantly correlated ($P<0.05$). Four of six correlations between the combined indices of biodiversity were significant, as were three of six correlations between the rarity measures. The correlation between the vulnerability measures was significant ($P<0.05$) only between amphibians and reptiles. Only amphibians and reptiles exhibited significant correlations for all criteria. The correlation coefficients were usually low.

The Standardised Biodiversity Index was highly correlated with the combined index of biodiversity of all *taxa* (r = 0.80 for amphibians, 0.64 for reptiles, 0.46 for nesting birds, and 0.51 for mammals, $P<0.0001$ in all cases). Thus, it was deemed a good synthetic diversity measure of species richness, rarity, and vulnerability of the four taxonomic groups together.

Congruence of high-value diversity areas

To what extent do areas of high-value diversity for the four *taxa* overlap? Despite most pairs of *taxa* exhibited non-random positive associations, congruence of areas of high-value diversity -cells within the 15% top segment of ranked values for the different criteria- among *taxa* was found to be moderate to low (Table 2.4, Fig. 2.2a-d). It was lowest for the pairs amphibians-birds, mammals-birds and mammals-reptiles based on species vulnerability (2.6% in all cases), and highest for the pair amphibians-reptiles based on species rarity (61.5%). Congruence averaged 40.5%, 47.8%, 16.2%, and 38.5% for areas of high-value diversity based on species richness, rarity, vulnerability, and the combined index of biodiversity, respectively. Thus, vulnerability is the criterion that produced the highest areas of high-value diversity dispersion for the different *taxa*, whereas rarity produced the highest areas of high-value diversity aggregation. The unconsistency of areas of high-value diversity congruence based on these two significantly correlated indices is explained by the low correlation coefficients between them for breeding birds and mammals (Table 2.2).

Evaluation of areas of high-value diversity

A few species were not included in the areas of high value diversity, irrespective of the criteria used to identify them (Table 2.5, Appendix 2.B). The performance based on the average proportion of species included in areas of high-value diversity for the different *taxa* were ranked: rarity (97.9%), combined index of biodiversity (97%), species richness (95.2%), and vulnerability (87.7%). The performance based on the average proportion of threatened species excluded from areas of high-value diversity for the different *taxa* were ranked: combined index of biodiversity and rarity (0.3%), vulnerability (9.9%), and species richness (13.8%). The areas of high-value diversity identified according to the Standardised Biodiversity Index of all *taxa* together included all amphibian and mammal species, all but 1 reptile species (categorized as rare) and all but 6 bird species (1 categorized as endangered, 2 categorized as rare, and the other 3 were non-threatened or introduced species) (Table 2.6, Appendix 2.B).

Coincidence between areas of high-value diversity and natural protected areas

A total of 110 cells included at least a fraction of a national, natural or regional park. The coincidence between areas of high-value diversity and cells including these natural protected areas was low (Table 2.7). Coincidences between areas of high-value diversity and natural protected areas were not statistically associated except for rarity of mammals and the combined index of bird diversity, which exhibited non-random positive associations. The coincidences between areas of high-value diversity based on the combined index of biodiversity and NPAs followed the rank amphibians < reptiles = mammals < birds. Thus, the existing network of natural protected areas provides a reasonable guarantee for conservation of bird diversity, whereas large pools of amphibian diversity are outside protected areas. Natural protected areas were included in 69.2% of areas of high-value diversity identified based on the Standardised Biodiversity Index of all *taxa* together (χ^2 = 13.46, P< 0.004) (Fig. 2.3).

Table 2.5. Numbers and percentages of species included and threatened species excluded from the areas of high-value diversity identifyed according to different criteria. Threatened species were defined according to Blanco & González (1992). Lists of species excluded are in Appendix 2.B.

	Amphibians		Reptiles		Breeding birds		Mammals	
	Species included	Threatened sp. excluded	Species included	Threatened sp. excluded	Species included	Threatened sp. excluded	Species included	Threatened sp. excluded
Richness	24 (88.8%)	2 (40%)	41 (97.6%)	1 (9%)	257 (95.5%)	5 (6.3%)	88 (98.9%)	0
Rarity	25 (92.6%)	0	42 (100%)	0	266 (98.9%)	1 (1.2%)	89 (100%)	0
Vulnerability	26 (96.3%)	0	39 (92.9%)	1 (9%)	223 (82.9%)	15 (19%)	70 (78.7%)	3 (11.5%)
Combined Index	26 (96.3%)	0	39 (92.9%)	0	266 (98.9%)	1 (1.2%)	89 (100%)	0

Table 2.6. Numbers and percentages -over the total number of species considered in this study- of species included in the areas of high-value diversity identified according to the Standardised Biodiversity Index of all taxa.

	Total number of species (427)	Number of endangered species (25)	Number of vulnerable species (34)	Number of rare species (45)	Number of undetermined species (17)	Number of insufficiently known species (28)
Amphibians	27(100%)	1 (100%)	1 (100%)	3 (100%)	0 (100%)	0 (100%)
Reptiles	41 (97.6%)	2 (100%)	4 (100%)	3 (75%)	1 (100%)	0 (100%)
Mammals	89 (100%)	6 (100%)	7 (100%)	5 (100%)	8 (100%)	9 (100%)
Breeding birds	263 (97.8%)	15 (93.8%)	22 (100%)	31 (94%)	8 (100%)	19 (100%)
All groups	420 (98.4%)	24 (96%)	34 (100%)	42 (93.3%)	17 (100%)	28 (100%)

Table 2.7. Coincidence in percentage between areas of high-value diversity and natural protected areas. The statistical significance of the associations have been tested by means of χ^2.

	Species richness	Rarity	Vulnerability	Combined index
Amphibians	43.8 %, χ^2 =0.27	53.9 %, χ^2 =2.39	56.4 %, χ^2 =3.6	53.8 %, χ^2 =2.43
Reptiles	61.4 %, χ^2 =6.8	56.4 %, χ^2 =2.49	56.4 %, χ^2 =3.6	61.5 %, χ^2 =6.83
Breeding birds	56 %, χ^2 =2.4	56.4 %, χ^2 =3.47	46.2 %, χ^2 =0.01	66.7 %, χ^2 =10.7**
Mammals	59 %, χ^2 =4.53	66.7 %, χ^2 =10.1*	35.9 %, χ^2 =3.02	61.5 %, χ^2 =6.12

*P <0.005, **P <0.01.

Discussion

Distribution of areas of high-value diversity

Within *taxa*, there were overall significant correlations between species richness, rarity, vulnerability, and a combined index of biodiversity, with the exception of richness and vulnerability, which were not correlated for any *taxa*, and richness and rarity, correlated just for mammals. This mismatch has been reported for many *taxa* in different parts of the world (Drinkrow & Cherry, 1995; Lombard, 1995; Williams *et al*., 1996; Fagan & Kareiva, 1997; Hacker *et al*., 1998; Fjeldsa, 2000). Conversely, Swengel (1998) found in Midwestern USA that within a habitat type, site rankings based on total numbers of butterfly species significantly tended to agree with site rankings based on numbers of specialist butterfly species. The combined index of biodiversity provided significant correlations with its components but "failed" in the cases of species richness of amphibians and vulnerability of mammals. More importantly, we found that among *taxa* the correlations between criteria

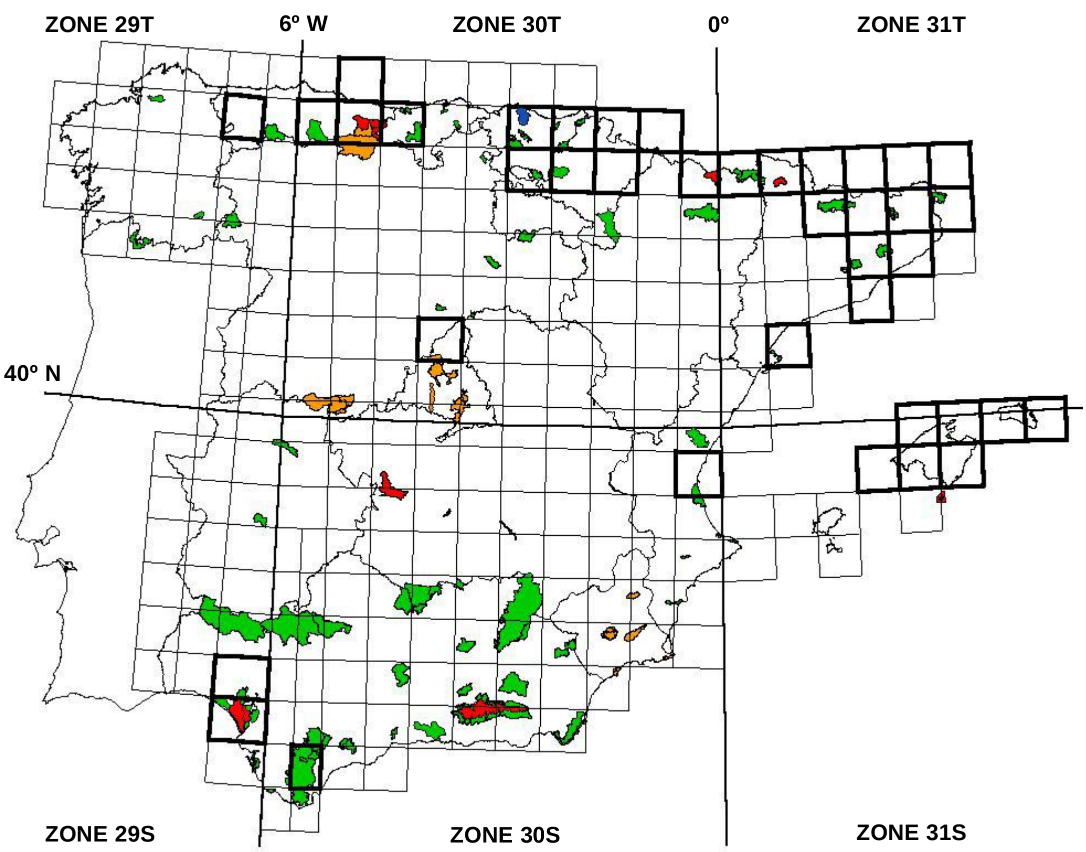


Figure 2.3. Coincidence of areas of high-value diversity identifyed according to the Standardised Biodiversity Index of all taxa together (units in bold type) and natural protected areas (symbol colors as in Fig. 2.1).

were very different. Species richness was always significantly correlated across *taxa*, but for vulnerability only amphibians and reptiles were correlated. These results indicate that, whereas there are certainly some macroecological patterns of species diversity that are consistent for different *taxa*, other determinants of the various components of species diversity differ. Overall, the balance leads to a moderate to low congruence between areas of high-value diversity identified according to different criteria and/or for different *taxa*. For analysis of congruence between *taxa* in other areas of the world, see Prendergast *et al*. (1993), Gaston *et al*. (1995), Lombard (1995), Balmford & Long (1996), Williams *et al*. (1997), Simmons *et al*. (1998), and Araujo (1999).

The effects of biogeography in areas of high-value diversity distribution are relevant, mostly for ectotherm species (Fig. 2.2). Biogeographic effects in Iberia include climate differences in the northern fringe of the Iberian Peninsula as compared to the rest of the country, and the insular effect in the Balearic Islands. Increased regional species diversity in climate transition zones is consistent with the analyses for plant species of Castro *et al*. (1997) and Rey Benayas & Scheiner (2002). These patterns may also be due, in part, to refugia in a land with over a millennium of varied agricultural, silvicultural, and pastoral practices. The peaks in species richness of amphibian, reptile and bird species in the central-western mountain ranges that we found (Fig. 2.2a-c) supports this interpretation. In the analysis of these authors, areas of high species richness were also rich in endemic species. At the scale of our analysis, we have found, though, a lack of relationship between species richness and rarity. Both ecotones and refugia may contribute to the higher regional species diversity of transitional zones.

Island biogeographic effects are also evident in our maps. These effects are very relevant for amphibian and reptile species (Fig. 2.2a,b), moderate for bird species (Fig. 2c), and inexistent for mammals (Fig. 2d). The explanations for these patterns may lie in the differences in dispersion capabilities and speciation and extinction rates of the *taxa* (Blondel & Aronson, 1999). Mammals are poor colonizers of islands, and humans drove to extinction all authoctonous mammal species in the Balearic Islands. Currently, all mammals except bats have been introduced by humans in these islands (Alcover *et al*., 1998; Palmer *et al*., 1999).

Besides biogeographic and refugia effects, patterns of areas of high-value diversity distribution have additional explanations that are relevant at regional and local scales. Differences in the ecological requirements for the *taxa* contribute to these patterns. Thus, a large proportion of areas of high-value diversity of amphibian species concentrate in northern Spain (Fig. 2.2a) since the higher precipitation and lower evaporation rates provide a greater moisture in air and soil as well as flooded habitats suitable for reproduction. Conversely, there is an aggregation of areas of high-value diversity of reptile species in the dry southern part of the Iberian Peninsula (Fig. 2.2b) (Meliadou and Troumbis, 1997; James and Shine, 2000). Large wetlands in the country are consistently associated to areas of high-value diversity of bird species, since many of them are specific of this kind of habitats. Fortunately, all large wetlands have legal protection (natural protected areas 6, 8, 45, 61 and others in Appendix 2.A, Fig. 2.1).

Identifying areas of high-value diversity as a tool for nature conservation

This study attempted to provide decision makers data needed to identify areas especially worthy of conservation (Prendergast *et al*., 1999), but our approach can be applied anywhere. We acknowledge that definitions of areas of high-value diversity are to some extent arbitrary, and our results are obviously scale dependent. A finer scale of resolution would likely result in a diminished areas of high-value diversity congruence among *taxa* (J.M. Rey Benayas, unpublished data). Overall, the areas of high-value diversity using the combined index of biodiversity had the best performance, capturing over 93% of the species and with fewer than 10% of threatened species uncaptured (Table 2.5). The greater effectiveness of the rarity criterion, as compared to the richness criterion, is in agreement with other studies (Haeupler & Vogel, 1999). Congruence of areas of high-value diversity of the different *taxa* was moderate to low. This makes conservation strategies more difficult.

Our Standardised Biodiversity Index of all *taxa* resulted in a site selection that included all amphibian and mammal species, all but one reptile species and all but six bird species (four out of these seven species are endangered, Table 2.6 and Appendix 2.B). Elsewhere in the world, where species distribution data are often sparse and mapped at even larger scales, and where decision making on reserve placement usually has to be very rapid, we suggest that a strategy based solely on one or two criteria, and on only one or a limited number of *taxa*, may fail to provide adequate protection for many organisms. The areas of high-value diversity approach could benefit from adding human threats (especially vegetation destruction) and land ownership and value to the geographical analysis of biodiversity (Dobson *et al*., 2001; Scott *et al*., 2001). Areas harboring high levels of species diversity and that are also under severe threat are usually defined in the literature as hotspots (Myers, 1988 and 1990; Prendergarst & Eversham, 1995; Beissinger *et al*., 1996; Harcourt, 2000; Myers *et al*., 2000).

In our study, when one looks at the coincidence between areas of high-value diversity according to the various criteria and natural protected areas, the scenario is discouraging (see Table 2.7). However, only 30% of areas of high-value diversity based on the Standardised Biodiversity Index of all *taxa* did not include any natural protected area (Fig. 2.3). In agreement with other authors, we suggest that this gap should be corrected urgently (Barnard *et al*., 1998; Simonetti, 1999). Spain's vertebrate fauna could be more effectively protected with only moderate acquisition of new, well-sited protected areas. While 70% of the areas of high-value diversity (modal area of 2500 km^2) included natural protected areas, these average only 274.6 km^2. NPAs occupy a small fraction of areas of high-value diversity, and there is no guarantee that the species found in a areas of high-value diversity site will be present in a fraction of natural protected area. Unfortunately, very few natural protected areas have complete lists of their species. Only for these sites is it possible to compare the diversity of protected areas and associated areas of high-value diversity. We urge that all natural protected areas be surveyed in order to strengthen nature conservation, and assess the effective long-term viability of the species in these areas (Hansen *et al*., 1993). Other studies on the coincidence between areas of high-value diversity and natural protected areas are those of Scott *et al*. (1993), Fearnside & Ferraz (1995), Lombard (1995), Jaffre *et al*. (1998), Savitsky & Lacher (1998), Araujo (1999), Maddock & Benn (2000), and Scott *et al*. (2001).

Biodiversity should not be the only target of conservation of natural protected areas (Gómez-Limón & de Lucio, 1995; Burger, 2000). Water and nutrient cycles, ecological processes such as productivity, and many other values and functions (landscape, geological, anthropological, aesthetical, spiritual, recreational, cultural) have been recognized in the scientific literature as important elements for conservation (Franklin, 1993; De Leo & Levin, 1997; Ettema *et al*., 1998; Rothley, 1999). Charismatic organisms such as some vertebrate groups may not always be the most appropriate surrogates for biodiversity (Prendergast *et al*., 1993; Araujo, 1999). This study did not consider either areas of high-value diversity for migratory, non-breeding birds, which add nearly 400 species to the avian fauna. The approach taken in this paper should be applied to as many other *taxa* as possible, in order to aid the formulation of the National Strategic Plan for Biodiversity Conservation. It is also useful for designing the Natura 2000 European Network of Special Conservation Areas, which pursues the preservation of representative and threatened habitats and species in the European Union (IUCN, 1994; European Commission, 1996).

Biodiversity conservation must not depend solely on natural protected areas. Management of traditional agricultural systems and forests are keystones for conservation outside natural protected areas (Hawthorne, 1996; Neitlich & McCune, 1997; Martin & Eadie, 1999; Loumou *et al*., 2000; Atauri & de Lucio, 2001). Maps of areas of high-value diversity may be useful for ecological restoration. It is possible to identify actions to be taken in areas of high-value diversity to foster the populations of the most endangered species. Some examples are habitat reconstruction (e.g. ponds), revegetation (e.g. riverine forests or island forests), refuge sites and food supply (e.g. rabbits for predators such as the lynx and the Iberian imperial eagle) (St. Clair *et al*., 1998; Brooker *et al*., 1999). These maps may also be useful for environmental impact assessment and mitigation (Ayensu *et al*., 1999). We overlapped a map of areas of high-value diversity and a map of natural protected areas. Next, we will overlap our map of areas of high-value diversity with the planned road, high-speed train railway and resevoir maps. Many recommendations will then arise. For instance, five large reservoirs are planned to be built in the Pyrenean Mountains, a zone of areas of high-value diversity concentration and important for migratory birds.

Margules & Pressey (2000) pointed to six stages in systematic conservation planning: 1) compile biodiversity data, 2) identify conservation goals, 3) review existing conservation areas, 4) select additional conservation areas, 5) implement conservation actions, and 6) maintain the required values of conservation areas. We took a first step for biodiversity conservation planning in the studied region, and point out the following conclusions from our study. Since the results are scale dependent, we advise that biodiversity data should be compiled with higher resolution, particularly for the natural protected areas (stage 1). For the goal of conservation of the largest pools of vertebrate diversity that takes into account rarity and vulnerability criteria (stage 2), our study has highlighted a gap in the existing natural protected area network (stage 3) within areas of high-value diversity. Their identification is the starting point to undertake stage 4 and set priorities related to stages 5 and 6 such as ecosystem management and restoration and environmental impact assessment and mitigation. The question is: will politicians finally pay attention to take notice of ecological research? (Brussard, 1991; Mooney, 1991). Time will tell.

Acknowledgments

This study was funded by the Determinantes de la diversidad biológica en ecosistemas mediterráneos. Papel de los procesos locales y regionales (CICYT AMB96-1161) and Factores limitantes de la revegetación con especies leñosas autóctonas de áreas degradadas en ambientes mediterráneos. Rendimiento de distintas actuaciones de manejo projects (CICYT REN2000-745). Europarc-España provided useful information about natural protected areas. We are indebted to José V. de Lucio, Juan Herrero, Jaime Potti, Salvador Rebollo, Sam Scheiner, Jon Paul Rodríguez, Bradford Hawkins, and an anonymous reviewer for their comments on an earlier version of this manuscript.

References

Alcover, J.A., Sans, A., Palmer, M., 1998. The extent of extinctions of mammals on islands. Journal of Biogeography 25, 913-918.

Araujo, M.B., 1999. Distribution patterns of biodiversity and the design of a representative reserve network in Portugal. Diversity and Distributions 5, 151-163.

Atauri, J.A., de Lucio, J.V., 2001. The role of landscape structure in species richness and distribution of birds, amphibians, reptiles and lepidopterans in Mediterranean landscapes. Landscape Ecology 16, 147-159.

Ayensu, E., Van R. Claasen, D., Collins, M., Dearing, A., Fresco, L., Gadgil, M., Gitay, H., Glaser, G., Juma, C., Krebs, J., Lenton, R., Lubchenco, J., McNeely, J.A., Mooney, H.A., Pinstrup-Andersen, P., Ramos, M., Raven, P., Reid, W.V., Samper, C., Sarukhán, J., Schei, P., Galízia Tundisi, J., Watson, R.T., Guanhua, X., Zakri, A.H., 1999. International ecosystem assessment. Science 286, 685-686.

Balmford, A., Long, A., 1996. Across-country analyses of biodiversity congruence with current conservation efforts in the tropics. Conservation Biology 9, 1539-1547.

Barnard, P., Brown, C.J., Jarvis, A.M., Robertson, A., Van-Rooyen, L., 1998. Extending the Namibian protected area network to safeguard hotspots of endemism and diversity. Biodiversity and Conservation 7, 531-547.

Beissinger, S.R., Steadman, E.C., Wohlgenant, T., Blate, G., Zack, S., 1996. Null models for assessing ecosystem conservation priorities: Threatened birds as titers of threatened ecosystems in South America. Conservation Biology 10, 1343-1352.

Bininda-Emonds, O.R.P., Vazquez, D.P., Manne, L.L., 2000. The calculus of biodiversity: Integrating phylogeny and conservation. Trends in Ecology and Evolution 15, 92-94.

Blanco, J.C., González, J.L., 1992. Libro Rojo de los Vertebrados de España. ICONA-Ministerio de Agricultura, Pesca y Alimentación. Madrid, Spain.

Blondel, J., Aronson, J., 1999. Biology and Wildlife of the Mediterranean Region. Oxford University Press, Oxford, UK.

Brooker, L., Brooker, M., Cale, P., 1999. Animal dispersal in fragmented habitat: measuring habitat connectivity, corridor use, and dispersal mortality. Conservation Ecology [online] 3(1): 4. Available from the Internet. URL: http://www.consecol.org/vol3/iss1/art4.

Brussard, P.F., 1991. The role of ecology in biological conservation. Ecological Applications 1, 6-12.

Burger, J., 2000. Landscapes, tourism, and conservation. Science of the Total Environment 249, 39-49.

Cameron, R.A.D., 1998. Dilemmas of rarity: Biogeographic insights and conservation priorities for land mollusca. Journal of Conchology SPECIAL PUBL. 2, 51-60.

Castro, I., Moreno, J.C., Humphries, C.J., Williams, P.H., 1997. Strengthening the Natural and National Park system of Iberia to conserve vascular plants. Botanical Journal of the Linnean Society 121, 189-206.

Ceballos, G., Rodríguez, P., Medellín R.A., 1998. Assessing conservation priorities in megadiverse Mexico: mammalian diversity, endemicity, and endangerment. Ecological Applications 8, 8-17.

Cincotta, R.P., Wisnewski, J., Engelman, R., 2000. Human population in the biodiversity hotspots. Nature 404, 990-992.

Costanza, R., Arge, R., de Groot,R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world´s ecosystem services and natural capital. Nature 387, 253-260.

De Leo, G.A., Levin S., 1997. The multifaceted aspects of ecosystem integrity. Conservation Ecology [online]1(1): 3. Available from the Internet. URL: http://www.consecol.org/vol1/iss1/art3.

Dobson, A.P., Rodríguez, J.P., Roberts, W.M., 2001. Synoptic tinkering: integrating strategies for large-scale conservation. Ecological Applications 11, 411-420.

Drinkrow, D.R., Cherry M.I., 1995. Anuran distribution, diversity and conservation in South Africa, Lesotho and Swaziland. South African Journal of Zoology 30, 82-90.

Ettema, C.H., Coleman, D.C., Vellidis, G., Lowrance, R., Rathbun, S.L., 1998. Spatiotemporal distributions of bacterivorous nematodes and soil resources in a restored riparian wetland. Ecology 79, 2721-2734.

European Commission. 1996. Interpretation manual of European Union Habitats. Version EUR 15. Brussels, Belgium.

Fagan, W.F., Kareiva, P.M., 1997. Using compiled species lists to make biodiversity comparisons among regions: A test case using Oregon butterflies. Biological Conservation 80, 249-259.

Faith, D.P., Walker, P.P.A., 1996. How do indicator groups provide information about the relative biodiversity of different sets of areas?: On hotspots, complementarity and pattern-based approaches. Biodiversity Letters 3, 18-25.

Fearnside, P.M., Ferraz, J., 1995. A conservation gap analysis of Brazil´s Amazonian vegetation. Conservation Biology 9, 1134-1147.

Fjeldsa, J., 2000. The relevance of systematics in choosing priority areas for global conservation. Environmental Conservation 27, 67-75.

Font Tullot, I., 1983. Atlas Climático de España. Instituto Nacional de Meteorología, Madrid, Spain.

Franklin, J.F., 1993. Preserving biodiversity: species, ecosystems, or landscapes? Ecological Applications 3, 202-205.

Gasc, J.P., Cabela, A., Crnobrnja-Isailovic, J., Dolmen, D., Grossenbacher, K., Haffner, P., Lescure, J., Martens, H., Martínez Rica, J.P., Maurin, H., Oliveira, M.E., Sofianidou, T.S., Veith, M., Zuiderwijk, A., editors, 1997. Atlas of Amphibians and Reptiles in Europe. Societas Europaea Herpetologica & Muséum National d´Histoire Naturelle (IEGB/SPN), Paris, France.

Gaston, K.J., Williams, P.H., Eggleton, P., Humphries, C.J., 1995. Large scale patterns of biodiversity: Spatial variation in family richness. Proceedings of the Royal Society of London Series B Biological Sciences 260, 149-154.

Gómez Limón, F.J., de Lucio J.V., 1995. Recreational activities and loss of diversity in grasslands in Alta Manzanares Natural Park, Spain. Biological Conservation, 74, 99-105.

Gómez Limón, F.J., de Lucio, J.V., Múgica, M., 2000. Los Espacios Naturales Protegidos del Estado Español en el umbral del siglo XXI. Fundación Fernando González Bernáldez. Madrid, Spain.

Groombridge, B., editor, 1992. Global Diversity: Status of the Earth´s Living Resources. Chapman and Hall, London, UK.

Hacker, J.E., Cowlishaw, G., Williams, P.H., 1998. Patterns of African primate diversity and their evaluation for the selection of conservation areas. Biological Conservation 84, 251-262.

Haeupler, H., Vogel, A., 1999. Plant diversity in Germany: A second review. Acta Botanica Fennica 162, 55-59.

Hagemeijer, W.J.M., Blair, M.J., editors, 1997. The EBCC Atlas of European Breeding Birds: Their Distribution and Abundance. T. and A.D. Poyser Ltd./European Bird Census Council. London, UK.

Hansen, A.J., Garman, S.L., Marks, B., Urban, D.L., 1993. An approach for managing vertebrate diversity across multiple-use landscapes. Ecological Applications 3, 481-496.

Harcourt, A.H., 2000. Coincidence and mismatch of biodiversity hotspots: A global survey for the order, primates. Biological-Conservation 93, 163-175.

Hawthorne, W.D., 1996. Holes and the sums of parts in Ghanaian forest: Regeneration, scale and sustainable use. Proceedings of the Royal Society of Edinburgh Section B (Biological Sciences) 104, 75-176.

Heywood, V.H., editor, 1995. Global Biodiversity Assessment. Cambridge University Press, Cambridge, UK.

IUCN, 1988. Red List of Threatened Animals. Gland, Switzerland.

IUCN, 1994. Parques para la Vida. Plan de Acción para las Áreas protegidas de Europa. Gland, Switzerland.

Jaffre, T., Bouchet, P., Veillon, J.M., 1998. Threatened plants of New Caledonia: Is the system of protected areas adequate? Biodiversity and Conservation 7, 109-135.

James, C.D., Shine, R., 2000. Why are there so many coexisting species of lizards in Australian deserts? Oecologia 125, 127-141.

James, A.N., Gaston, K.J., Balmford, A., 1999. Balancing the Earth´s accounts. Nature 401, 323-324.

Lombard, A.T. 1995. The problems with multi-species conservation: Do hotspots, ideal reserves and existing reserves coincide? South African Journal of Zoology 30, 145-163.

Loumou, A., Giourga, C., Dimitrakopoulos, P., Koukoulas, S., 2000. Tourism Contribution to Agro-Ecosystems Conservation: The Case of Lesbos Island, Greece. Environmental Management 26, 363-370.

Maddock, A., Benn, G.A., 2000. Identification of conservation-worthy areas in northern Zululand, South Africa. Conservation Biology 14, 155-166.

Margules C.R., Pressey, R.L. 2000. Systematic conservation planning. Nature 405, 243-253.

Martin, K., Eadie, J.M., 1999. Nest webs: A community-wide approach to the management and conservation of cavity-nesting forest birds. Forest Ecology and Management 115, 243-257.

Meliadou, A., Troumbis, A.Y., 1997. Aspects of heterogeneity in the distribution of diversity of the European herpetofauna. Acta Oecologica 18, 393-412.

Mitchell-Jones, A.J., Amori, G., Bogdanowicz, W., Krystufek, B., Reijnders, P.J.H., Spitzenberger, F., Stubbe, M., Thissen, J.B.M., Vohralík, V., Zima, J., 1999. The Atlas of European Mammals. T. and A.D. Poyser Ltd/Societas Europaea Mammalogica. London, UK.

Mittermeier, R.A., Myers, N., Thomsen, J.B., Da Fonseca, G.A.B., Olivieri, S., 1998. Biodiversity hotspots and major tropical wilderness areas: Approaches to setting conservation priorities. Conservation Biology 12, 516-520.

Mooney, H.A., 1991. Emergence of the study of global ecology: Is terrestrial ecology an impediment to progress? Ecological Applications 1, 2-5.

Myers, N., 1988. Threatened biotas: "hotspots" in tropical forests. Environmentalist 8, 187-208.

Myers, N., 1990. The biodiversity challenge: expanded hotspot analysis. Environmentalist 10, 243-256.

Myers, N., Mittermeler, R.A., Mittermeler, C.G., da Fonseca, G.A.B., Kent J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853-858.

Neitlich, P.N, McCune, B., 1997. Hotspots of epiphytic lichen diversity in two young managed forests. Conservation Biology 11, 172-182.

Palmer, M., Pons, G.X., Cambefort, I., Alcover, J.A., 1999. Historical processes and environmental factors as determinants of inter-island differences in endemic faunas: the case of the Balearic Islands. Journal of Biogeography 26, 813-823.

Perring, F.H., Farrel, L., 1983. British Red Data Books: 1. Vascular Plants. RSNC, Nenleham, UK.

Pimentel, D., Wilson, C., McCullum, C., Huang, R., Dwen, P., Flack, J., Tran, Q., Saltman, T., Cliff, B., 1997. Economic and environmental benefits of biodiversity. BioScience 47, 747-757.

Pimm, S.L., Raven, P., 2000. Extinction by numbers. Nature 403, 843-844.

Prendergast, J.R., Eversham, B.C., 1995. Butterfly diversity in southern Britain: Hotspot losses since 1930. Biological Conservation 72, 109-114

Prendergast, J.R., Quinn, R.M., Lawton, J.H., Eversham, B.C., Gibbons, D.W., 1993. Rare species, the coincidence of diversity hotspots and conservation strategies. Nature 365, 335-337.

Prendergast, J.R., Quinn, R.M., Lawton, J.H., 1999. The gaps between theory and practice in selecting nature reserves. Conservation Biology 13, 484-492.

Rabinowitz, D., 1981. Seven forms of rarity. Pages 205-217 in H. Synge, editor. The biological aspects of rare plant conservation. John Wiley, Chichester, UK.

Raven, P.H., Wilson, E.O., 1992. A fifty-year plan for biodiversity surveys. Science 258, 1099-1100.

Reid, W.V., 1998. Biodiversity hotspots. Trends in Ecology and Evolution 13, 275-280.

Rey Benayas, J.M., Scheiner, S.M., García Sánchez-Colomer, M., Levassor, C., 1999. Commonness and rarity: theory and application of a new model to Mediterranean montane grasslands. Conservation Ecology [online] 3(1):5. Available from the Internet. URL: http://www.consecol.org/vol3/iss1/art5.

Rey Benayas, J.M., Scheiner, S.M., 2002. Plant diversity, biogeography, and environment in Iberia: patterns and inferred mechanisms. Journal of Vegetation Science 13, 245-258.

Ricketts, T.H., Ricketts, T.H., Dinerstein, E., Olson, D.M., Loucks, C.J., Eichbaum, W., DellaSala, D., Kavanagh, K., Hedao, P., Hurley, P.T., Carney, K.M., Abell, R., Walters. S., 1999. Terrestrial

Ecoregions of North America: a Conservation Assessment. Island Press, Washington, USA.

Rothley, K.D., 1999. Designing bioreserve networks to satisfy multiple, conflicting demands. Ecological Applications 9, 741-750.

Savitsky B.G., Lacher, Jr, T.E., (eds.), 1998. GIS methodologies for developing conservation strategies: tropical forest recovery and wildlife management in Costa Rica. Columbia University Press, New York, USA.

Scheiner, S.M., (1992) Measuring pattern diversity. Ecology 73, 1860-1867.

Scott, J.M., Davis, F., Csuti, B., Noss, R., Butterfield, B., Groves, C., Anderson, H., Caicco, S., D'Erchia, F., Edwards, Jr., T.C., Ulliman, J., Wrigth, R.G., 1993. Gap analysis: a geographical approach to protection of biological diversity. Wildlife Monograph No. 123, The Wildlife Society.

Scott, J.M., Davis, F.W., McGhie, R.G., Wright, R.G., Groves, C., Estes, J., 2001. Nature reserves: do they capture the full range of America´s biological diversity? Ecological Applications 11, 999-1007.

SEO/Birdlife, 1997. Atlas de las Aves de España (1975-1995). Lynx Edicions, Barcelona, Spain.

Simmons, R.E., Griffin, M., Griffin, R.E., Marais, E., Kolberg, H., 1998. Endemism in Namibia: Patterns, Processes and predictions. Biodiversity and Conservation 7, 513-530.

Simonetti, J.A., 1999. Diversity and conservation of terrestrial vertebrates in mediterranean Chile. Revista Chilena de Historia Natural 72, 493-500.

Soulé, M.E., Sanjayan, M.A. 1998. Conservation targets: Do they help? Science 279, 2060-2061.

St. Clair, C.C., Bélisle, M., Desrochers, A., Hannon, S., 1998. Winter responses of forest birds to habitat corridors and gaps. Conservation Ecology [online] 2(2): 13. Available from the Internet. URL: http://www.consecol.org/vol2/iss2/art13.

Swengel, A.B., 1998. Comparisons of butterfly richness and abundance measures in prairie and barrens. Biodiversity and Conservation 7, 1639-1659.

Terborgh, J., 1999. Requiem for Nature. Island Press, Washington, USA.

Tilman, D., 1999. The ecological consequences of changes in biodiversity: a search for general principles. Ecology 80, 1455-1474.

Usher, M.B., editor, 1986. Wildlife Conservation Evaluation. Chapman and Hall, London, UK.

Virolainen, K.M., Nättinen, K., Suhonen,J., Kuitunen, M., 2001. Selecting herb-rich forest networks to protect different measures of biodiversity. Ecological Applications 11, 411-420.

Williams, P.H., Humphries, C.J., Vane-Wright, R.L., 1991. Measuring biodiversity: taxonomic relatedness for conservation priorities. Australian Systematic Botany 4, 665-679.

Williams, P., Gibbons, D., Margules, C., Rebelo, A., Humphries, C., Pressey, R., 1996. A comparison of richness hotspots, rarity hotspots, and complementary areas for conserving diversity of British birds. Conservation Biology 10, 155-174.

Williams, P.H., Gaston, K.J., Humphries, C.J., 1997. Mapping biodiversity value worldwide: Combining higher-taxon richness from different groups. Proceedings of the Royal Society of London Series B Biological Sciences 264, 141-148.

Appendix 2.A. List of national, natural and regional parks that were considered in this study. Numbers match Fig. 2.1. Source: Europarc-España database.

<u>National Parks</u> (IUCN category II): 1. Picos de Europa, 2. Ordesa y Monte Perdido, 3. Aigüestortes i Estany de Sant Maurici, 4. Cabañeros, 5. Tablas de Daimiel, 6. Archipiélago de Cabrera, 7. Doñana, 8. Sierra Nevada.

<u>Natural Parks</u> (IUCN category V): 9. Fragas do Eume, 10. Complexo Dunar de Corrubedo e Lagoas de Carregal e Vixán, 11. Islas Cíes, 12. Monte Aloia, 13. Baixia Limia-Serra do Xurés, 14. O Invernadeiro, 15. Somiedo, 16. Redes, 17. Oyambre, 18. Dunas de Liencres (Piélagos), 19. Saja-Besaya, 20. Macizo de Peña Cabarga, 21. Collados del Asón, 22. Urdabai (Biosphere Reserve), 23. Área de Gorbeia, 24.Urkiola, 25. Aralar, 26. Valderejo, 27. Pagoeta, 28. Izki, 29. Aiako Harria, 30. Señorío de Bértiz, 31. Urbasa y Andía, 32. Bardenas Reales, 33. Sierra de Cebollera, 34. Moncayo, 35. La Sierra y Cañones de Guara, 36. Posets-Maladeta, 37. Cadí-Moixeró, 38. Zona Volcánica de la Garrotxa, 39. Aiguamolls de l'Empordá, 40. Cap de Creus, 41. Montseny, 42. Sant Llorenç del Munt i l'Obac, 43. Montserrat, 44. Delta de l'Ebre, 45. Lago de Sanabria y alrededores, 47. Cañon del Río Lobos, 48. Hoces del Río Duratón, 50. Monfragüe, 51. Cornalvo, 52. Cumbre, Circo y Lagunas de Peñalara, 56. Hayedo de Tejera Negra, 57. Lagunas de Ruidera, 58. Prat de Cabanes-

Torreblanca, 59. Sierra Espadán, 60. La Albufera, 61. Marjal de Pego-Oliva, 62. El Montgó, 63. Penyal d'Ifac, 64. Carrascal de la Font Roja, 65. El Hondo, 66. Salinas de Santa Pola, 67. Sa Dragonera, 68. S'Albufera de Mallorca, 69. Mondragó, 70. L'Albufera des Grao, la illa d'en Colom i el cap de Favàritx, 71. Sierra de Aracena y Picos de Aroche, 72. Sierra Norte de Sevilla, 73. Sierra de Hornachuelos, 74. Sierra de Cardeña y Montoro, 75. Sierra de Andujar, 76. Despeñaperros, 77. Sierras de Cazorla, Segura y las Villas, 78. Sierra de Castril, 79. Entorno de Doñana, 80. Sierras Subbéticas, 81. Sierra de Huétor, 82. Sierra Mágina, 83. Sierra de Baza, 84. Sierra de María-Los Vélez, 85. Bahía de Cádiz, 86. La Breña y Marismas del Barbate, 87. Sierra de Grazalema, 88. Los Alcornocales, 89. Sierra de las Nieves, 90. Montes de Málaga, 91. Sierras de Tejeda, Almijara y Alhama, 92. Sierra Nevada, 93. Cabo de Gata-Níjar.

<u>Regional Parks</u> (IUCN category V): 46. Picos de Europa, 49. Sierra de Gredos, 53. Cuenca Alta del Río Manzanares, 54. Curso Medio del Río Guadarrama y su entorno, 55. Entorno de los ejes de los cursos bajos de los Ríos Manzanares y Jarama, 94. Sierra de La Pila, 95. Sierra Espuña, 96. Carrascoy y El Valle, 97. Cabo de Cope-Puntas de Calnegre, 98. Salinas y Arenales de San Pedro, 99. Calblanque, Monte de Las Cenizas y Peña del Aguila.

Appendix 2.B. Species excluded from the high-value diversity areas identified according to different criteria and their vulnerability status according to the *Red Book of Spanish Vertebrates* (Blanco & González 1992). Abbreviations are the following: E = endangered, V = vulnerable, R = rare, U = undetermined, I-K = insufficiently known, I = introduced, and N-T = non-threatened.

<u>According to species richness</u>. **Amphibians**: *Alytes muletensis* (E), *Bufo viridis* (R), and *Hydromantes ambrosii*; **reptiles**: *Podarcis lilfordi* (V); **breeding birds**: *Amandava amandava* (I), *Bucanetes githagineus* (N-T), *Estrilda melpoda* (I), *Falco eleonorae* (R), *Numenius arquata* (R), *Pandion haliaetus* (E), *Plegadis falcinellus* (N-T), *Psittacula krameri* (I), *Puffinus mauretanicus* (N-T), *Rissa tridactyla* (R), *Sylvia sarda* (N-T), and *Uria aalge* (E); **mammals**: *Ammotragus lervia* (I).

<u>According to rarity</u>. **Amphibians**: *Alytes cisternasii* (N-T), and *Peurodeles waltl* (N-T); **breeding birds**: *Amandava amandava* (I), *Aythya fuligula* (N-T), and *Corvus frugilegus* (R).

<u>According to vulnerability</u>. **Amphibians**: *Alytes cisternasii* (N-T); **reptiles**: *Algyroides marchi* (R), *Lacerta monticola* (N-T), and *Lacerta schreiberi* (N-T); **breeding birds**: *Acrocephalus schoenobaenus* (N-T), *Aegolius funereus* (R), *Anthus pratensis* (N-T), *Anthus spinoletta* (N-T), *Anthus trivialis* (N-T), *Aythya fuligula* (N-T), *Bucanetes githagineus* (N-T), *Carduelis spinus* (N-T), *Certhia familiaris* (N-T), *Circus cyaneus* (I-K), *Corvus frugilegus* (R), *Crex crex* (U), *Cygnus olor* (I), *Charadrius morinellus* (R), *Dendrocopos leucotos* (E), *Dendrocopos medius* (V), *Dryocopus martius* (R), *Emberiza citrinella* (N-T), *Estrilda melpoda* (I), *Fidecula hypoleuca* (N-T), *Gallinago gallinago* (I-K), *Gypaetus barbatus* (E), *Lagopus mutus* (E), *Lanius collurio* (N-T), *Lanius minor* (E), *Locustella naevia* (N-T), *Luscinia svecica* (N-T), *Montifringilla nivalis* (N-T), *Myiopsitta monachus* (I), *Numenius arquata* (R), *Parus palustris* (N-T), *Perdix perdix* (V), *Phalacrocorax carbo* (N-T), *Phylloscopus sibilatrix* (I-K), *Phylloscopus trochilus* (N-T), *Plegadis falcinellus* (N-T), *Prunella collaris* (N-T), *Psittacula krameri* (I), *Pyrrhocorax graculus* (N-T), *Rissa tridactyla* (R), *Saxicula rubetra* (N-T), *Scolopax rusticola* (I-K), *Tetrao urogallus* (V), *Tichodroma muraria* (N-T), *Turdus torquatus* (N-T), and *Uria aalge* (E); **mammals**: *Ammotragus lervia* (I), *Apodemus flavicollis* (N-T), *Arvicola terrestris* (N-T), *Chionomys nivalis* (N-T), *Glis glis* (N-T), *Lepus castroviejo* (R), *Lepus europaeus* (N-T), *Marmota marmota* (I), *Martes martes* (N-T), *Micromys minutus* (N-T), *Microtus gerbei* (N-T), *Mustela lutreola* (E), *Myocastor coypus* (N-T), *Rupicapra pyrenaica* (N-T), *Sorex araneus* (N-T), *Sorex coronatus* (N-T), *Sorex minutus* (N-T), *Talpa europaea* (N-T), and *Ursus arctos* (E).

<u>According to the combined index of biodiversity</u>. **Amphibians**: *Alytes cisternasii* (N-T); **reptiles**: *Lacerta monticola* (N-T), *Lacerta schreiberi* (N-T), and *Vipera seoanei* (N-T); **breeding birds**: *Aythya fuligula* (N-T), *Corvus frugilegus* (R), and Estrilda melpoda (I).

<u>According to the standardised biodiversity index of all taxa</u>. **Reptiles**: *Algyroides marchi* (R); **breeding birds**: *Amandava amandava* (I), *Bucanetes githagineus* (N-T), *Corvus frugileus* (R), *Estrilda melpoda* (I), *Rissa tridactyla* (R), and *Uria aalge* (E).

Capítulo 3

El mejor bioindicador para un paisano de que algo malo pasa es que el cuco no anuncie sus fiestas de mayo: "Si el pecu no canta, pal veinte de abril, o se ha muerto el pecu, o viene el fin"

Lamina de B. Lwerker que ilustra algunos mamíferos endémicos de Tasmania. *The Geographical Distribution of Animals*. Alfred R. Wallace. 1876.

Capítulo 3

Identificación de áreas relevantes de diversidad de herpetofauna que están amenazadas por proyectos de infraestructuras planeados en España

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

Jose M. Rey Benayas, Enrique de la Montaña, Josabel Belliure, Xavier R. Eekhout (2006). Identifying areas of high herpetofauna diversity that are threatened by planned infrastructure projects in Spain. Journal of Environmental Management 79, 279-289.

Resumen

Una importante tarea, relacionada con la conservación, es predecir si los proyectos de infraestructuras planeados pueden amenazar la biodiversidad. Es este estudio se evalúa el potencial impacto de las infraestructuras planeadas en España sobre las especies de anfibios y reptiles, dos grupos altamente vulnerables debido a su limitada capacidad de dispersión y a la actual situación de declive poblacional. Se utilizaron datos de distribución de ambos grupos para identificar áreas relevantes de diversidad de herpetofauna, y se compararon con la localización de las autovías y autopistas, las líneas férreas de alta velocidad y los embalses planeados. Se usaron cuatro criterios para identificar esas áreas: riqueza de especies, rareza, vulnerabilidad, y un índice que combina esos tres criterios. A partir de un total de 1441 celdas de 20 x 20 km, fueron definidas las áreas relevantes como aquellas celdas que ordenadas según el valor de los distintos criterios conseguían incluir todas las especies y todas las especies amenazadas. El índice combinado necesita el menor número de celdas para incluir todas las especies amenazadas (1.7 y 2.6% de las celdas para anfibios y reptiles, respectivamente). La coincidencia entre estas áreas relevantes de diversidad y las celdas que incluyen infraestructuras planeadas -denominadas "unidades de alerta para la planificación"- fue del 35.4% para anfibios y del 31.2% para reptiles. La mitigación de los potenciales impactos debería incluir acciones como la construcción de barreras para el acceso de los animales a las carreteras y vías férreas, y ecoductos bajo estas construcciones. Nuestro trabajo proporciona a las autoridades encargadas de la conservación información que puede ser empleada para tomar decisiones que protejan los hábitats. Una técnica que identifica amenazas en la herpetofauna antes de que ocurran, probablemente también aumentará las posibilidades de que la herpetofauna sea protegida.

Palabras clave: Conservación; Herpetofauna; Índice combinado de biodiversidad; Impacto ambiental potencial; Unidades de alerta para la planificación

Identifying areas of high herpetofauna diversity that are threatened by planned infrastructure projects in Spain

José M. Rey-Benayas[1], Enrique de la Montaña[1], Josabel Belliure[1], Xavier R. Eekhout[2]

[1] *Departamento de Ecología, Universidad de Alcalá. Alcalá de Henares, 28871 Madrid, Spain.*

[2] *Museo Nacional de Ciencias Naturales, CSIC, 28006 Madrid, Spain.*

Abstract

A major task related to conservation is to predict if planned infrastructure projects are likely to threaten biodiversity. In this study we investigated the potential impact of planned infra structure in Spain on amphibian and reptile species, two highly vulnerable groups given their limited dispersal and current situation of population decline. We used distribution data of both groups to identify areas of high herpetofauna diversity, and compared the location of these areas with the location of the planned road, high-speed train railway and water reservoir network. Four criteria were used for this identification: species richness, rarity, vulnerability, and a combined index of the three criteria. From a total of 1441 cells of 20 x 20 km, areas of high diversity were defined as those cells whose ranked values for the different criteria included either all species or all threatened species. The combined index provided the smallest number of cells needed to retain all threatened species (1.7% and 2.6% of the cells for amphibian and reptile species, respectively). Coincidences between these high diversity areas and cells including planned infrastructures -denominated "alert planning units"- were 35.4% for amphibians and 31.2% for reptiles. Mitigation of such a potential impact would include actions as barriers to animal access to roads and railways and ecoducts under these constructions. Our approach provides conservation authorities information that can be used to make decisions on habitat protection. A technique that identifies threats to herpetofauna before they occur is also likely to improve the chance of herpetofauna being protected.

Keywords: Alert planning units; Combined index of biodiversity; Conservation; Herpetofauna; Potential environmental impact

Introduction

An intense debate within the scientific community on biodiversity conservation strategies over the last decade has been fed by increasing rates of biodiversity loss (Heywood, 1995; Costanza *et al*., 1997; Pimentel *et al*., 1997; Ricketts, 1999; Terborgh, 1999; Tilman, 1999; Bininda-Emonds *et al*., 2000; Cincotta *et al*., 2000; Myers *et al*., 2000; Pimm & Raven, 2000; Dietz & Adger, 2003; Mittermeier *et al*., 2003). Identifying areas with outstanding biodiversity features helps provide information that decision makers can use, together with other information such as cost, to determine priorities for conservation (Pearlstine *et al*., 2002; Sarkar & Margules, 2002; Williams *et al*., 2002; Matsuda *et al*., 2003; Rey Benayas & de la Montaña, 2003; Rodrigues *et al*., 2004). These features are most frequently pinpointed based on criteria such as species richness, rarity (particularly endemic *taxa*), taxonomic uniqueness, threatened species, representativeness, and indicator *taxa* (Kirkpatrick, 1983; Usher, 1986; Williams *et al*., 1991; Prendergast *et al*., 1993; Faith & Walker, 1996; Castro *et al*., 1997; Reid, 1998; Rey Benayas *et al*., 1999; Williams & Araujo, 2000; Virolainen *et al*., 2001; Garson *et al*., 2002; Margules *et al*., 2002).

Establishing protected areas seems to be one of the most useful tools for preserving large pools of biodiversity, and constitutes the cornerstone on which regional strategies are built (Margules & Pressey, 2000; Gaston *et al*., 2002; Williams *et al*., 2002). Many studies have addressed the issue of identifying priority areas for conservation for gap analysis purposes, i.e. the detection of highly valuable areas that do not include nature reserves. However, reserves alone are not enough for nature conservation. The speed of anthropogenic change is accelerating and has dramatically increased the risk of habitat loss and disturbance (Corbett, 1989; Adams, 1999; Krzysciak, 2000; Kiesecker *et al*., 2001; Faith & Walker, 2002). Therefore, an important task is foreseeing potential threats to particular species, groups of species, and valuable sites for conservation such as planned infrastructure. This issue has received little attention in the scientific literature.

Our aims are, first, to determine if planned infrastructure projects are likely to threaten herpetofauna in Spain. Secondly, we asked which species could be most affected by these infra structures and which areas are needed to maintain all amphibian and reptile species free of infrastructure impacts. One of our specific targets is to achieve set of areas that contain (i) all species and (ii) all threatened species. Finally, we make the point that areas of overlap of high herpetological diversity and infrastructure should be monitored by conservationists. These identified "alert planning units" should be considered candidate areas for actions to mitigate environmental impact. Planned infrastructure in Spain includes the construction of about 5000 km of highways, 2000 km of high-speed train railways, and 100 water reservoirs. More than 90% of the Spanish territory is not legally protected and is thus susceptible to affectation by such new infrastructure. A technique that identifies threats to herpetofauna before they occur is also likely to improve the chance of herpetofauna being protected.

We used amphibians and reptiles as target groups because they are two *taxa* that are highly vulnerable due to their current population declines and ecological requirements (Blaustein *et al*., 1997; Lips, 1998; Pounds *et al*., 1999; Houlahan *et al*., 2000; Kiesecker *et al*., 2001; Green, 2003). Most of these species have small home ranges and are sedentary. They are usually poor colonizers, and are often reliant on a brief immature phase for dispersal. With a few exceptions, these species exhibit very limited long-distance movement capabilities for dispersal over a large region. They therefore have little capacity to avoid even temporary threats or changes to their habitats (Corbett, 1989; Adams, 1999; Krzysciak, 2000; Kiesecker *et al*., 2001; Biek *et al*., 2002; Green, 2003). Amphibians and reptiles are threatened by habitat loss, land use change, and in many cases human antipathy (Corbett,1989; Krzysciak, 2000; Semlitsch, 2000; Biek *et al*., 2002). Reports of declining amphibian populations in many parts of the world are numerous, particularly in the last few decades, and are attributable to factors such as habitat destruction and fragmentation, increased road density and traffic, alien predators, contaminants, emerging infection diseases, and climatic change (Gardner, 2001; Stow *et al*., 2001; Collins & Storfer, 2003; Kats & Ferrer, 2003). Some of the above mentioned factors affect local populations, whereas others may have more widespread impact (Davenport, 1997; Richter *et al*., 1997; Adams, 1999; Kolozsvary & Swihart, 1999; Rouse *et al*., 1999; Gibbons *et al*., 2000; Krzysciak, 2000; Semlitsch, 2000; Cohen, 2001).

We used four criteria for identifying areas of high herpetofauna diversity: species richness, rarity, vulnerability, and a combined index of the three criteria. Next, we evaluated the efficiency of the various criteria used to identify these areas. Finally, we compared the location of these areas with the location of the infrastructure projects, and identified the coincidences as alert planning units. Our intention is to provide useful information for herpetofauna conservation. "Alert maps" may be useful to decision-makers because they point where in the country large pools of amphibian and reptile diversity are particularly under threat and proper actions can be taken. Our analyses are illustrative, not exhaustive. A similar approach can be used either for different species groups, criteria, or threats to biodiversity.

Materials and Methods

The study area

Spain is one of the richest countries in the European Union with respect to amphibian and reptile diversity, with 28 and 58 species, respectively. The study area includes the Spanish fraction of the Iberian Peninsula and the Balearic Islands (Fig. 3.1). It embraces a variety of biomes, relief, climates, and soil types despite a relatively small area (585644 km^2 in total). Two major climatic zones, Mediterranean and Atlantic, are present (Font Tullot, 1983). The Mediterranean climate, with its seasonality, warm, dry summers and cool, wet winters, is characteristic of most of Iberia and the Balearic Islands. The Atlantic climate is wetter, cooler and less seasonal and is found in a band ca. 100 km wide along the western and northern coast and also influences the Pyrenean Mountains in the northeast. The driest and warmest areas in the south of the country served as refuges during Pleistocene glaciations. In contrast, the northern transition zone between the two climates is substantially younger, having emerged only after glacial retreat. Within regions, the relative extent of different vegetation types, natural landscapes, and diversity patterns depend not only on the environmental status and variation, but also on human impacts. Thus, land management -particularly agriculture- can affect diversity (Leiva *et al*., 1997).

Planning units and data sources

Our analyses used cells of 20 x 20 km, defined by UTM coordinates, as planning units. We examined the presence and absence of amphibian and reptile species in 1441 cells. We built the cell-by-species matrices using the species distribution maps from Pleguezuelos (1997). We considered 28 amphibian species and 48 non-marine reptile species (Appendix 3.A).

Criteria for identifying priority areas of high herpetofauna diversity

We used four criteria to identify areas of high herpetofauna diversity: species richness, rarity, vulnerability, and a combined index of the three criteria. There are many forms of rarity, responding to different combinations of geographical range, local abundance, habitat specificity, and habitat occupancy (Rabinowitz, 1981; Rey Benayas *et al*., 1999). In this study, rarity of a species *i* was defined by its geographical range measured as the inverse of the number of cells where it was present ($1/n_i$). Currently, there are not established criteria in Spain classifying species into rarity categories according to their geographical ranges (Perring & Farrel, 1983, Cameron, 1998). For a cell r, the rarity index was $\sum_{i=1}^{s}(1/n_{ri})/s_r$, where s_r was the number of species found in the cell.

Species vulnerability was quantified using the categories of the *Red Book of Spanish Vertebrates* (Blanco & González, 1992, Appendix 3.A). The species categories that were considered are the following: endange-

Figure 3.1. Map of continental Spain and the Balearic Islands. It illustrates the planned infrastructure network considered in this study. Symbols: solid lines are highways, dashed lines are high-speed railways, and gray squares are reservoirs.

red, vulnerable, rare, undetermined, insufficiently known, introduced, and non-threatened. These categories were previously defined by the International Union for Conservation of Nature (IUCN 1988). Complete definitions can be found in Appendix 3.A. These categories are now under revision. Vulnerability is actually a surrogate concept of rarity plus rates of habitat loss and other threats. We assigned every category a score related to its degree of vulnerability, ranging from 5 for endangered species to 1 for non-threatened and introduced species. Intermediate categories were assigned 4 (vulnerable and undetermined), 3 (rare) and 2 (insufficiently known). We acknowledge the subjectivity of these scores; they represent a rank and thus a relative value. For a cell, the vulnerability index was $\sum_{i=1}^{s} V_{ri}/s_r$, where V_{ri} was the vulnerability score of the species *i* present in the cell. Finally, we used a combined index of species richness, rarity, and vulnerability defined by Rey Benayas & de la Montaña (2003): $\sum_{i=1}^{s}(1/n_{ri})V_{ri}$. In this index, species richness is implicit in $\sum_{i=1}^{s}$.

Next, all diversity indices for both *taxa* were ranked. To quantitatively define areas of high herpetofauna diversity, we considered the pool of cells within the upper ranked values for the various criteria that included either all species or all threatened species. For our purposes, "threatened species" were considered those belonging to the endangered, vulnerable, rare, and undetermined categories of IUCN (1988).

The planned infrastructure network

We obtained information on the locations of newly planned highways and roads, high-speed train railways, and water reservoirs (Fig. 3.1) till year 2007 from official public documents available on-line at http://www.mfom.es/home/Infraes/intro.html.

Data analysis

We examined the relationships between the four criteria across *taxa* by means of correlation analysis using Bonferroni corrections for multiple comparisons. To evaluate the effectiveness of the various criteria used to identify areas of high diversity, we looked at the number of ranked cells that included all species and all threatened species. The congruence between areas of high diversity for both *taxa* was analyzed by means of χ^2. Then, we examined the coincidence between the location of these areas and the location of the planned infrastructure. Those cells that were categorized as areas of high diversity and that included planned infrastructure were considered alert planning units. Next, we ascertained which species were present only in alert planning units and how many of the cells occupied by threatened species within the areas of high diversity were alert planning units. Finally, when necessary, we calculated the number of additional cells without planned infrastructures that should be added to the selected areas of high diversity to ensure the representativeness goal (retention of all species).

Results

Evaluation and distribution of areas of high herpetofauna diversity

The performance of the four indices based on the average number of cells needed to retain all species for both *taxa* were ranked: rarity (75 cells) < vulnerability (122) < combined index (159.5 cells) < species richness (626.5) (Table 3.1). However, the number of cells based on the combined index drops to 49 if the endemic *Alytes dickhilleni* is removed from the analysis. The performance of the indices based on the average number of cells needed to retain all threatened species for both *taxa* were ranked: combined index (30.5 cells) < rarity (46.5) < vulnerability (57.5) < species richness (626.5) (Table 3.1). All cells where the five threatened amphibian species were present were encompassed by the combined index. Similarly, all cells where the thirteen threatened reptile species appeared were retained by this index, with the exception of 82 out of 109, 5 out of 46, and 7 out of 58 cells for *Emys orbicularis*, *Coluber viridiflavus*, and *Elaphe longissima*, respectively. Thus, the combined index performed better than the other criteria because fewer cells were needed to retain threatened herpetofauna diversity.

Distribution of areas of high diversity for amphibians and reptiles as defined by the combined index is shown in Fig. 3.2. These areas for amphibians are mainly aggregated in the Atlantic climatic region of Iberia and in the Balearic Islands (Fig. 3.2a). The distribution of areas of high reptile diversity indicates an aggregation in the Balearic Islands, Pyrenean Mountains, and the southern coast (Fig. 3.2b). The interior section of the Iberian Peninsula is less favored for both taxonomic groups.

Congruence of areas of high amphibian and reptile diversity

The correlation coefficients between each criterion used to identify areas of high diversity between the two *taxa* were 0.73, 0.13, 0.35, and 0.05 for richness, rarity, vulnerability and the combined index, respectively (n=1441, P<0.0001 in all cases except for the combined index which was not significant at P=0.05 after correcting for multiple comparisons). To what extent do areas of high diversity for both *taxa* overlap? Using the results shown in Table 3.1 and Fig. 3.2, congruence between these areas for amphibians and reptiles averaged 43.3% for all criteria. The vulnerability criterion produced the highest dispersion of areas of high diversity for both *taxa* (18.9% congruence, χ^2=15.86, P=0.0012), whereas richness produced the highest aggregation (87.9% congruence, χ^2=356.6, P<0.0001). Rarity (32.7%, χ^2=64.0, P<0.0001) and the combined index of biodiversity (33.8%, χ^2=54.14, P<0.0001) produced an intermediate level of congruence.

Table 3.1. Number (and proportion in parenthesis) of cells that are needed to retain all species and threatened species of amphibians and reptiles according to four criteria.

	All amphibian species	Threatened amphibian species	All reptile species	Threatened reptile species
Richness	850 (59.0%)	850 (59.0%)	505 (35.0%)	505 (35.0%)
Rarity	95 (6.6%)	38 (2.63%)	55 (3.8%)	48 (3.3%)
Vulnerability	115 (8.0%)	47 (3.3%)	128 (8.9%)	67 (4.6%)
Combined index	130 (9.02%)	24 (1.7%)	189 (13.1%)	37 (2.6%)

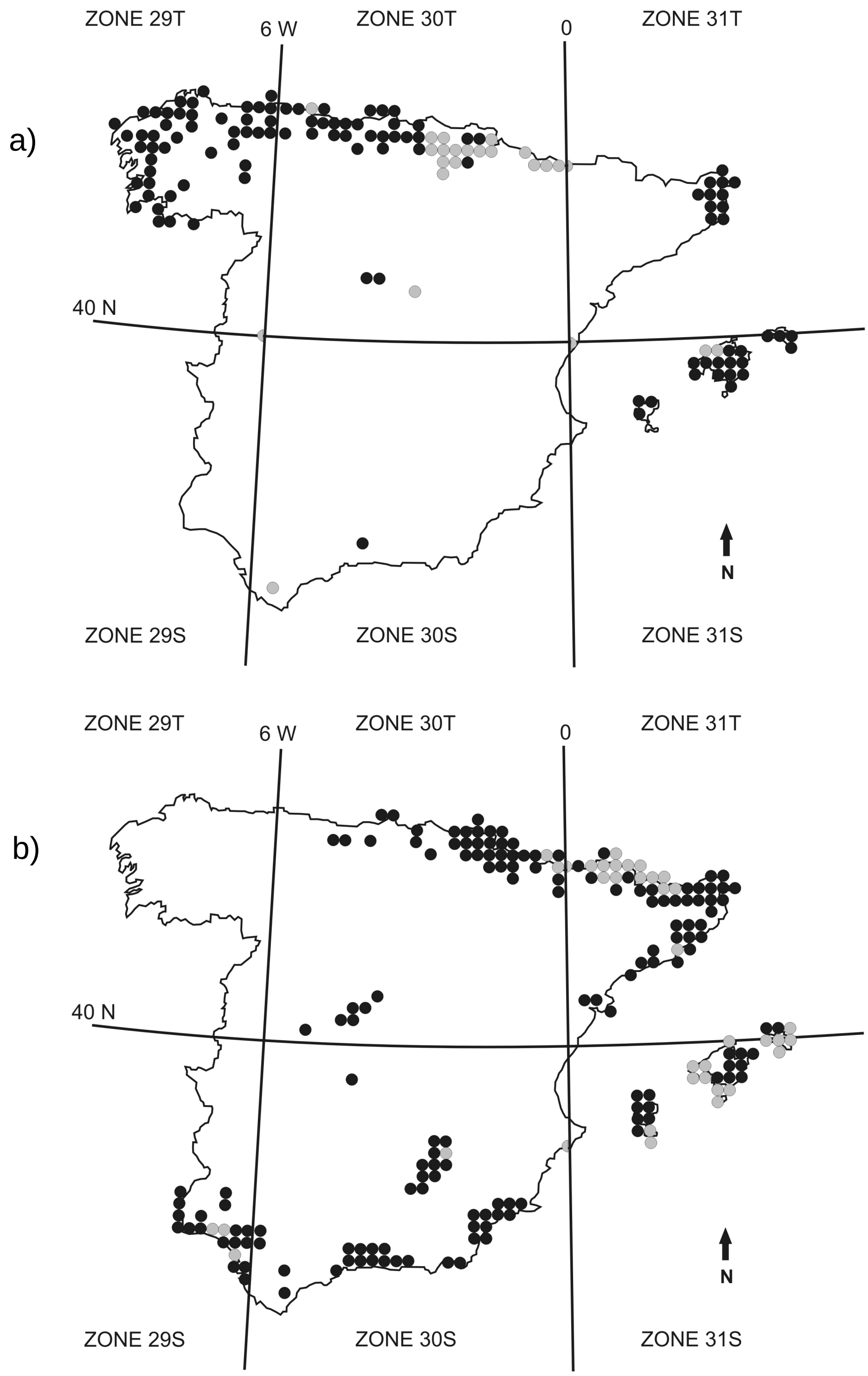


Figure 3.2. Maps of areas of high diversity identified according to the combined index of biodiversity for a) amphibians (top) and b) reptiles (bottom). Symbols are the following: solid circles retain all species and gray circles retain all threatened species.

Coincidence between areas of high herpetofauna diversity and planned infrastructure

Coincidences between areas of high diversity and cells including planned infrastructure were low and not statistically associated. Forty six (35.4%) and 59 (31.2%) cells identified as areas of high diversity according to the combined index for amphibians and reptiles, respectively, were affected by planned infrastructures (denominated alert planning units in Fig. 3.3). Only one amphibian species (the non-native toad *Bufo mauritanicus*) and one reptile species (the non-native turtle *Pseudemys picta*) were exclusive to alert planning units. Since these non-native, introduced species appeared in only one cell, we did not need to examine which cells free of planned infrastructure should be added to the areas of high diversity list to retain the species lost in the alert sites. However, five threatened amphibian species lost a substantial presence in this list if alert planning units were eliminated: *Chioglossa lusitanica* (from 47 to 25 cells), *Triturus alpestris* (41 to 24), *Bufo viridis* (21 to 17), *Rana dalmatina* (12 to 7), and *Alytes muletensis* (2 to 1). Planned infrastructure did not affect any cell where five threatened reptile species were present (*Lacerta agilis*, *L. araica*, *L. aureolioi*, *Podarcis lilfordi*, and *P. pityusensis*), while the remaining threatened reptile species were affected by the infrastructure, with the following loss of cells where they were present: *Coluber viridiflavus* (from 41 to 32 cells), *Elaphe longissima* (51 to 33), *Testudo hermanni* (34 to 21), *Emys orbicularis* (27 to 17), *Chamaleo chamaleon* (30 to 21), *T. graeca* (21 to 12), *Algyroides marchi* (11 to 7), and *Lacerta bonnali* (7 to 5).

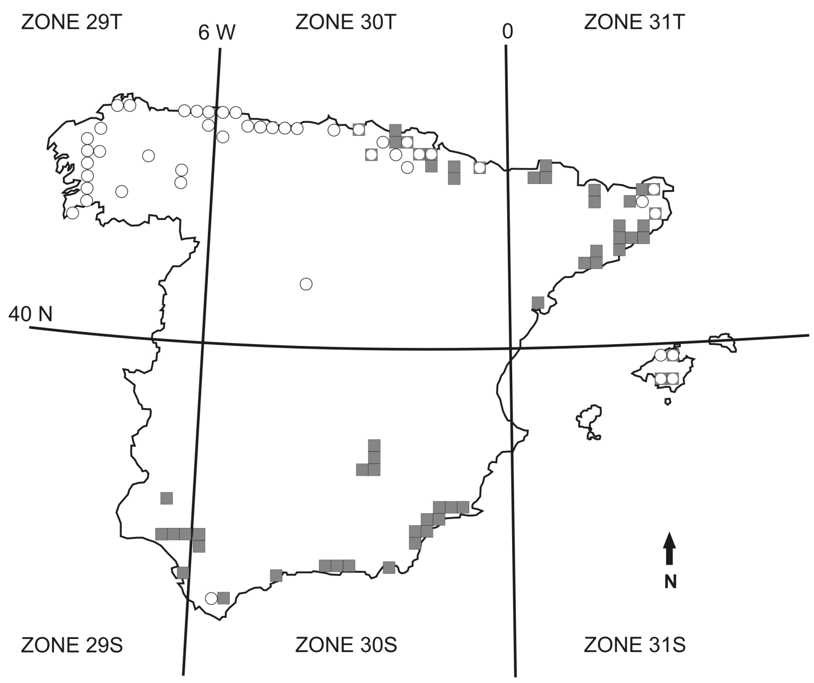


Figure 3.3. Map of alert planning units (i.e., coincidences between areas of high diversity identified according to the combined index of biodiversity and planned infrastructure). Symbols: empty circles are alert planning units for amphibian species and gray squares are alert planning units for reptile species.

Discussion

Distribution of areas of high herpetofauna diversity

The number of species in any region is likely to depend on its location (Davidowitz & Rosenzweig, 1998). In the European context, a clear pattern of species numbers and ratios of amphibians to reptiles emerges from north to south. The northward territories have fewer species, with a higher proportion of amphibians, and this holds true even for islands (Corbett, 1989). Diversity of amphibians and reptiles in our study area supports this pattern.

Three groups of ecological and evolutionary factors important for the distribution of areas of high amphibian and reptile diversity are biogeography, the effects of mountain refuges, and the ecological requirements of the species. Biogeographic effects in the Iberian Peninsula include: 1) climate differences in the northern fringe as compared to the rest of the Peninsula, particularly the transition between the Atlantic and Mediterranean climates, and 2) the insular effect in the Balearic Islands. Increased regional species diversity in transition zones is consistent with the analyses for plant species of Rey Benayas & Scheiner (2002). The Iberian Peninsula includes a high proportion of amphibian (25%) and reptile (20%) endemics. Many species that are characteristic of the Atlantic climate are only present at the northern part of Iberia, and its origin is presumably the refuge effect of Iberia for the European fauna during the Quaternary (Vargas & Real, 1997). By contrast, the Strait of Gibraltar notably exerted a barrier effect (Busak, 1986). As a consequence, the Iberian herpetofauna has higher similarity with northwestern Europe than with Northern Africa (Oosterbroek & Arntzen, 1992). The peaks in species richness of amphibian and reptile species found in the central-western mountain ranges and other areas such as the Cantabrian and Pyrenean mountains may also be due to refuge in a land with over a millennium of varied agricultural, silvicultural, and pastoral practices. This explanation was favored by Castro *et al*. (1997) for the distribution of terrestrial vascular plants in the same study area. Island biogeographic effects are also evident in our maps. Most of the territory of the Balearic Islands shows areas of high diversity. The explanations for these patterns may lie in the differences in dispersion capabilities and speciation and extinction rates of the *taxa* (Blondel & Aronson, 1999).

Apart from biogeographic and *refugia* effects, differences in the ecological requirements of the *taxa* also contribute to the patterns of the distribution of areas of high diversity (Meliadou & Troumbis, 1997, James & Shine, 2000). A large proportion of these areas for amphibian species are concentrated in northern Spain as higher precipitation and lower evaporation rates increase moisture in air and soil as well as flooded habitats suitable for reproduction. Conversely, there is an aggregation of areas of high reptile diversity in the dry southern part of the Iberian Peninsula. The difference in ecological requirements for both taxonomic groups leads to a moderate to low congruence between their respective areas of high diversity.

Anthropogenic factors influencing the patterns of species diversity should be considered as well. A study on biogeographical regions of the Iberian Peninsula, based on the distribution of freshwater fish and amphibians, assessed the influence of humans based upon data of native and well-established introduced species

(Vargas *et al*., 1998). The effect of species introductions by humans is clearer in islands. The presence of 13 amphibian and reptile species non-endemic to the Balearic Islands is the result of anthropogenic introductions during the nearly 8000 years of sea traffic between the islands and the continent (Mayol, 1997).

Efficiency of the criteria used to define areas of high diversity

Our results show differences in the effectiveness of the different criteria used to define areas of high diversity. Rarity and the combined index showed the highest efficiency since fewer cells were needed to retain all species or all threatened species. The combined index of Rey Benayas & de la Montaña (2003) has the additional value of simultaneously taking into account species richness, geographical rarity, and vulnerability and allowed the retention of most of the cells where the threatened species were present. Richness showed the lowest efficiency as many cells were needed to retain significant pools of diversity. This fact is important, as species richness constitutes one of the most utilized criterions in conservation decisions (Caldecott *et al*, 1996; Rossi & Kuitunen, 1996; Médail & Quézel, 1997; Reyers *et al*., 2000; Pearlstine *et al*., 2002; Rodrigues *et al*., 2004). The greater efficiency of the rarity criterion, as compared to the richness criterion, is supported by other studies (Williams *et al*., 1996; Margules *et al*., 1988; Haeupler & Vogel, 1999).

Areas of high diversity and planned infrastructure

Areas harboring high levels of species diversity and that are also under severe threat are usually defined in the literature as hotspots (Myers, 1988, 1990; Prendergarst & Eversham, 1995; Beissinger *et al*., 1996; Harcourt, 2000; Myers *et al*., 2000). We used diversity of two ecologically contrasting *taxa* that are highly vulnerable and that have historically been under-considered in conservation plans. Indeed, effective conservation measures remain inadequate as compared to other vertebrates and very few action plans for conservation of endangered species are currently being implemented in Spain (Márquez, 2004). The moderate to low congruence of areas of high diversity of both *taxa* makes decisions on conservation strategies more difficult.

The coincidences between areas of high diversity and the newly planned infrastructure are highest for road construction due to the higher spatial extent of this as compared to other types of infrastructure considered in our study. Road construction is a serious threat to biodiversity due to a variety of effects such as restricted movement between populations, increased mortality (particularly as the ecological requirements of ectotherms make roads an optimal site for basking), habitat fragmentation, greater edge effects, increased human access to wildlife habitats, and increased accessibility for exotic predators (May & Norton, 1996; Findlay & Bourdages, 2000). Populations of susceptible species are expected to decline gradually after road construction, with local extinction occurring sometime later. Thus, the full effects of road construction on these *taxa* may be undetectable for decades. Direct mortality (road kills) has been documented for both groups, representing 23 and 89% of the total vertebrate individuals killed in the study area to date (Barbadillo & García-Paris, 1991).

Other factors complicate assessment of the environmental impacts of roads, such as their distance to target populations. The density of paved roads on lands up to 2 km away from the habitat occupied by different species has been shown to influence species richness (Findlay & Houlahan, 1997). This suggests that most existing policies, which focus almost exclusively on actions within the habitat itself and/or a narrow buffer zone around the perimeter, are unlikely to provide adequate protection for biodiversity.

The lower level of coincidence of high-speed train railways with areas of high diversity responds to the reduced area potentially affected by this type of infrastructure in the study region. Their negative effects, however, can be very important in terms of fragmentation, and with the exception of the risk of mortality due to road-kill, largely coincide with those related to road construction.

The potential effects of newly planned reservoirs on the species have received little study. Water availability in the habitat is one of the most important factors affecting the temporal distribution of reproductive activities in amphibian species. However, reservoirs often imply canalization of small waterways and the loss of water levels in related water complexes that negatively affect their use by amphibians. Moreover, the walls of reservoirs and canals are often too steep and high to allow individuals access as an alternative habitat for reproduction (Barbadillo *et al*., 1997). The importance of lakeshore development on amphibian abundance has been studied by Woodford & Meyer (2003). Usually, far from being an advantage, these constructions represent a risk for species conservation.

There were several limitations of our study that should be considered in evaluating our results. We evaluated diversity from distribution maps that may have false absences. Given the available data, the definitions of areas of high diversity are to some extent arbitrary, and our results are obviously scale dependent. Some authors such as Pressey & Nicholls (1989) have criticized scoring approaches, but the results that we obtained are actually promising. Our approach would have benefited from adding other groups of species, additional human threats such as land-use or land-cover change, as well as land ownership and value to the geographical analysis of biodiversity and planned infrastructure (Dobson *et al*., 2001; Scott *et al*., 2001). It can also be argued there was a mismatch of scales and a "knowledge-action gap" (Pfeffer & Sutton, 1999), since we analyzed diversity at the grain of 20 x 20 km and most environmental impact assessment (EIA) and mitigation decisions address areas within 100´s of meters of a proposed development. Smaller cells will increase efficiency (Pressey & Logan, 1998), and might also reduce the number of alert planning units. This study has addressed biodiversity retention (*sensu* Cowling, 1999). The inclusion of some measure of biodiversity persistence (Araújo & Williams, 2000), or designing the network of areas of high diversity to incorporate environmental processes (e.g. Cowling *et al*., 2003a,b) would also considerably improve this assessment.

However, we believe that the common application of EIA at the project level fails to ensure adequate consideration of potentially serious trans-boundary, widespread, indirect, cumulative, and synergistic ecological effects (Treweek *et al*., 1998). Maps of areas of high diversity have been suggested as useful tools for environmental impact assessment and mitigation (Ayensu *et al*., 1999). Clearly, some form of strategic ecological assessment is required to ensure that the development of new infrastructure is compatible with the conservation of habitats and species. Our study highlights areas where planned infrastructures are likely to impact herpetofauna. Mitigation of such a potential impact would include actions such as barriers to animal access to roads and railways and ecoducts under these constructions (Joly *et al*., 2003; Kats & Ferrer, 2003; Rosell, 2003; Semlitsch & Bodie, 2003; Willson & Dorcas, 2003). Collaboration with stakeholders, especially those organizations proposing and undertaking these new infrastructures, and the development of an implementation strategy would greatly facilitate conservation interventions in alert planning units (Driver *et al*., 2003).

Conclusions

We produced a map that combines the location of areas of high herpetofauna diversity and the location of the planned increases in public infrastructure in Spain. Portions of the territory as small as 1.7 and 6.6% of the total area were found to include all threatened species and all species, respectively, with the additional value that these areas retain most of the cells occupied by the threatened species. The map highlighted a number of alert planning units that can be used in subsequent analysis by decision makers. We were able to identify which species will likely be the most affected by this infrastructure. Fortunately, we found that there is no need to identify additional sites free of planned infrastructure that would retain some lost species. This approach can be used to favor the conservation of other taxonomic groups anywhere in the world.

Acknowledgments

This study was funded by the projects REN2000/745 and CGL2004-00355/BOS (Ministerio de Educación y Ciencia, Spain) and GR/AMB/0757/2004 (Comunidad de Madrid). We are indebted to Salvador Rebollo, Rafael Márquez and Robert Manson for their comments on an earlier version of this manuscript. Four anonymous reviewers improved the contents and presentation of the current manuscript.

References

Adams, M.J., 1999. Correlated factors in amphibian decline: Exotic species and habitat change in western Washington. Journal of Wildlife Management 63, 1162-1171.

Araújo, M.B., Williams, P.H., 2000. Selecting areas for species persistence using occurrence data. Biological Conservation 96, 331-345.

Ayensu, E., Claasen, D. van R., Collins, M., Dearing, A., Fresco, L., Gadgil, M., Gitay, H., Glaser, G., Juma, C., Krebs, J., Lenton, R., Lubchenco, J., McNeely, J.A., Mooney, H.A., Pinstrup-Andersen, P., Ramos, M., Raven, P., Reid, W.V., Samper, C., Sarukhán, J., Schei, P., Galízia Tundisi, J., Watson, R.T., Guanhua, X., Zakri, A.H., 1999. International ecosystem assessment. Science 286, 685-686.

Barbadillo, L.J., Garcia-Paris, M., 1991. Problemas de conservación de los anfibios en España. Quercus 62, 20-25.

Barbadillo, L.J., Garcia-Paris, M., Sanchíz, B., 1997. Orígenes y relaciones evolutivas de la herpetofauna ibérica. In: Pleguezuelos, J.M. (Ed.), Distribución y biogeografía de los anfibios y reptiles en España y Portugal, Monografías de Herpetología 3, Universidad de Granada, Asociación Herpetológica Española, pp. 47-100.

Beissinger, S.R., Steadman, E.C., Wohlgenant, T., Blate, G., Zack, S., 1996. Null models for assessing ecosystem conservation priorities: Threatened birds as titers of threatened ecosystems in South America. Conservation Biology 10, 1343-1352.

Biek, R., Funk, W.C., Maxell, B.A., Mills L.S., 2002. What is missing in amphibian decline research: Insights from ecological sensitivity analysis. Conservation Biology 16, 728-734.

Bininda-Emonds, O.R.P., Vazquez, D.P., Manne, L.L.,

2000. The calculus of biodiversity: Integrating phylogeny and conservation. Trends in Ecology and Evolution 15, 92-94.

Blanco, J.C., González, J.L., 1992. Libro Rojo de los Vertebrados de España. ICONA-Ministerio de Agricultura, Pesca y Alimentación. Madrid, Spain.

Blaustein, A.R., Kiesecker, J.M., Chivers D.P., Anthony, R.G., 1997. Ambient UV-B radiation causes deformities in amphibian embryos. Proceedings of the Natural Academy of Science USA 94, 13735-13737.

Blondel, J., Aronson, J., 1999. Biology and Wildlife of the Mediterranean Region. Oxford University Press, Oxford, UK.

Busack, S.D., 1986. Biogeographic analysis of the herpetofauna separated by the formation of the Strait of Gibraltar. National Geographic Research 2, 17-36.

Caldecott, J.O., Jenkins, M.D., Johnson T.H., Groombridge, B., 1996. Priorities for conserving global species richness and endemism. Biodiversity and Conservation 5, 699-727.

Cameron, R.A.D., 1998. Dilemmas of rarity: Biogeographic insights and conservation priorities for land mollusca. Journal of Conchology Special Publication 2, 51-60.

Castro, I., Moreno, J.C., Humphries, C.J., Williams, P.H., 1997. Strengthening the Natural and National Park system of Iberia to conserve vascular plants. Botanical Journal of the Linnean Society 121, 189-206.

Cincotta, R.P., Wisnewski,, J., Engelman, R., 2000. Human population in the biodiversity hotspots. Nature 404, 990-992.

Cohen, M.M.Jr., 2001. Frog decline, frog malformations, and a comparison of frog and human health. American Journal of Medical Genetics 104, 101-109.

Collins, J.P., Storfer, A., 2003. Global amphibian declines: Sorting the hypotheses. Diversity and Distributions 9 (2), 89-98.

Corbett, K., 1989. Conservation of European Reptiles and Amphibians. Christopher Helm Publishers. Bromley, Kent. UK.

Costanza, R., Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world´s ecosystem services and natural capital. Nature 387, 253-260.

Cowling, R.M., 1999. Planning for persistence - systematic reserve design in Africa´s Succulent Karoo desert. Parks 9, 17-30.

Cowling, R.M., Lombard, A.T., Rouget, M., Kerley, G.I.H., Wolf, T., Sims-Castley, R., Knight, A.T., Vlok, J.H.J., Pierce, S.M., Boshoff, A.F., Wilson, S.L. 2003a. A Conservation Assessment for the Subtropical Thicket Biome. Terrestrial Ecology Research Unit Report No. 43, University of Port Elizabeth, South Africa.

Cowling, R.M., Pressey, R.L., Rouget, M., Lombart, A.T. 2003b. A conservation plan for a global biodiversity hotspot - the Cape Floristic Region, South Africa. Biological Conservation 112, 191-216.

Davenport, J., 1997. Temperature and the life-history strategies of turtles. Journal of thermal biology 22, 479-488.

Davidowitz, G., Rosenzweig M.L., 1998. The latitudinal gradient of species diversity among North American grasshoppers (Acrididae) within a single habitat: A test of the spatial heterogeneity hypothesis. Journal of Biogeography 25, 553-560.

Dietz, S., Adger, W.N., 2003. Economic growth, biodiversity loss and conservation effort. Journal of Environmental Management 68, 23-35.

Dobson, A.P., Rodríguez, J.P., Roberts, W.M., 2001. Synoptic tinkering: integrating strategies for large-scale conservation. Ecological Applications 11, 411-420.

Driver, A., Cowling, R.M., Maze, K. 2003. Planning for Living Landscapes: Perspectives and Lessons from South Africa. Center for Applied Biodiversity Science at Conservation International and the Botanical Society of South Africa, Washington, D.C., U.S.A. and Cape Town, South Africa.

Faith, D.P., Walker, P.A., 1996. How do indicator groups provide information about the relative biodiversity of different sets of areas?: On hotspots, complementarity and pattern-based approaches. Biodiversity Letters 3, 18-25.

Faith, D.P., Walker, P.A., 2002. The role of trade-offs in biodiversity conservation planning: linking local management, regional planning and global conservation efforts. Journal of Bioscience 27, 393-407.

Findlay, C.S., Bourdages, J., 2000. Response time on wetland biodiversity to road construction on adjacent lands. Conservation Biology 14, 86-94.

Findlay, C.S., Houlahan J., 1997. Anthropogenic correlates of species richness in southeastern Ontario wetlands. Conservation Biology 11, 1000-1009.

Font Tullot, I., 1983. Atlas Climático de España. Instituto Nacional de Meteorología, Madrid, Spain.

Gardner, T., 2001. Declining amphibian populations: A global phenomenon in conservation biology. Animal Biodiversity and Conservation 24 (2), 25-44.

Garson, J., Aggarwal, A., Sarkar, S., 2002. Birds as surrogates for biodiversity: an analysis of a data set from southern Quebec. Journal of Bioscience 27, 347-360.

Gaston, K.J., Pressey, R.L., Margules, C.R., 2002. Persistence and vulnerability: retaining biodiversity in the landscape and in protected areas. Journal of Bioscience 27, 361-384.

Gibbons, J.W., Scott, D.E., Ryan, T.J., Buhlmann, K.A., Tuberville, T.D., Metts, B.S., Greene, J.L., Mills, T., Leiden, Y., Poppy, S., Winne, C.T., 2000. The global decline of reptiles, deja vu amphibians. Bioscience 50, 653-666.

Green, D.M., 2003. The ecology of extinction: Population fluctuation and decline in amphibians. Biological Conservation 111, 331-343.

Haeupler, H., Vogel, A., 1999. Plant diversity in Germany: A second review. Acta Botanica Fennica 162, 55-59.

Harcourt, A.H., 2000. Coincidence and mismatch of biodiversity hotspots: A global survey for the order, primates. Biological Conservation 93, 163-175.

Heywood, V.H., 1995. Global Biodiversity Assessment. Cambridge University Press, Cambridge, UK.

Houlahan, J.E., Findlay, C.S., Schmidt, B.R., Meyer, A.H., Kuzmin, S.L., 2000. Quantitative evidence for global amphibian population declines. Nature 404, 752-755.

IUCN. 1988. Red List of Threatened Animals. Gland, Switzerland.

James, C.D., Shine, R., 2000. Why are there so many coexisting species of lizards in Australian deserts? Oecologia 125, 127-141.

Joly, P., Morand, C., Cohas, A., 2003. Habitat fragmentation and amphibian conservation: building a tool for assessing landscape matrix connectivity. Comptes rendus biologies 326, S1, 32-39.

Kats L.B., Ferrer R.P., 2003. Alien predators and amphibian declines: Review of two decades of science and the transition to conservation. Diversity and Distributions 9 (2), 99-110.

Kiesecker, J.M., Blaustein, A.R., Belden, L.K., 2001. Complex causes of amphibian population declines. Nature 410, 681-684.

Kirkpatrick, J.B., 1983. An iterative method for establishing priorities for the selection of nature reserves: an example from Tasmania. Biological Conservation 25, 127-134.

Krzysciak, K.R., 2000. What threatens amphibians at dawn of the new millennium? Wiadomosci Ekologiczne 46, 115-126.

Kolozsvary, M.B., Swihart, R.K., 1999. Habitat fragmentation and the distribution of amphibians: Patch and landscape correlates in farmland. Canadian Journal of Zoology 77, 1288-1299.

Leiva, M.J., Chapin III, F.S., Fernández Alés, R., 1997. Differences in species composition and diversity among Mediterranean grasslands with different history - the case of California and Spain. Ecography 20, 97-106.

Lips, K.R., 1996. Decline of a tropical montane amphibian fauna. Conservation Biology 12, 106-117.

Margules, C.R., Pressey, R.L., 2000. Systematic conservation planning. Nature 405, 243-253.

Margules, C.R., Nicholls, A.O., Pressey, R.L., 1988. Selecting networks of reserves to maximize biological diversity. Biological Conservation 43, 63-76.

Margules, C.R., Pressey,R.L., Williams, P.H., 2002. Representing biodiversity: data and procedures for identifying priority areas for conservation. Journal of Bioscience 27, 309-326.

Márquez, R., 2004. La conservación de los anfibios y reptiles en la España de las autonomías. Quercus 221, 28-35.

Matsuda, H., Serizawa, S., Ueda, K., Kato,T., Yahara,T., 2003. Assessing the impact of the Japanese 2005 World Exposition Project on vascular plants' risk of extinction. Chemosphere 53, 325-336.

May, S.A., Norton, T.W., 1996. Influence of fragmentation disturbance on the potential impact of feral predators on native fauna in Australian forest ecosystems. Wildlife Research 23, 387-400.

Mayol, J., 1997. Biogeografía de los anfibios y reptiles de las islas baleares. In: Pleguezuelos, J.M. (Ed.), Distribución y biogeografía de los anfibios y reptiles en España y Portugal, Monografías de Herpetología 3, Universidad de Granada, Asociación Herpetológica Española, pp. 371-379.

Medail, F., Quézel, P., 1997. Hot-Spots analysis for Conservation of plant biodiversity in the Mediterranean Basin. Annales Missouri Botanical Garden 84, 112-127.

Meliadou, A., Troumbis, A.Y., 1997. Aspects of heterogeneity in the distribution of diversity of the European herpetofauna. Acta Oecologica 18, 393-412.

Mittermeier, R.A., Mittermeier, C.G., Brooks, T.M., Pilgrim, J.D., Konstant, W.R., da-Fonseca, G.A., Kormos, C., 2003. Wilderness and biodiversity conservation. Proceedings of the National Academy of Science USA 100, 10309-10313.

Myers, N., 1988. Threatened biotas: "hotspots" in tropical forests. Environmentalist 8, 187-208.

Myers, N., 1990. The biodiversity challenge: expanded hotspot analysis. Environmentalist 10, 243-256.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853-858.

Oosterbroek. P., Arntzen, J.W., 1992. Area-Cladograms of Circum-Medoterranean *taxa* in relation to Mediterranean palaeogeography. Journal of Biogeography 19, 3-20.

Pearlstine, L.G., Smith, S.E., Brandt, L.A., Allen, C.R., Kitchens, W.M., Stenberg, J., 2002. Assessing state-wide biodiversity in the Florida Gap analysis project. Journal of Environmental Management 66(2), 127-144.

Perring, F.H., Farrel, L., 1983. British Red Data Books: 1. Vascular Plants. RSNC, Nenleham, UK.

Pfeffer, J., Sutton, R.I.,1999. Knowing "What" to do is not enough: turning knowledge into action. California Management Review 42, 83-107.

Pimentel, D., Wilson, C., McCullum, C., Huang, R., Dwen, P., Flack, J., Tran, Q., Saltman, T., Cliff, B., 1997. Economic and environmental benefits of biodiversity. BioScience 47, 747-757.

Pimm, S L., Raven, P., 2000. Extinction by numbers. Nature 403, 843-844.

Pleguezuelos, J.M., 1997. Distribución y biogeografía de los anfibios y reptiles en España y Portugal. Universidad de Granada, Granada, España.

Pounds, J.A.,. Fogden, M.P.L., Campbell, J.H., 1999. Biological response to climate change on a tropical mountain. Nature 398, 611-615.

Prendergast, J.R., Eversham, B.C., 1995. Butterfly diversity in southern Britain: Hotspot losses since 1930. Biological Conservation 72, 109-114.

Prendergast, J.R., Quinn, R.M., Lawton, J.H., Eversham, B.C., Gibbons, D.W., 1993. Rare species, the coincidence of diversity hotspots and conservation strategies. Nature 365, 335-337.

Pressey, R.L., Logan, V.S., 1998. Size of selection units for future reserves and its influence on actual vs targeted representation of features: a case study in Western New South Wales. Biological Conservation 85, 305-319.

Pressey, R.L., Nicholls, A.O., 1989. efficiency in conservation evaluation: scoring vs. iterative approaches. Biological Conservation 50, 199-218

Rabinowitz, D., 1981. Seven forms of rarity. In: Synge, H., (Ed.), The biological aspects of rare plant conservation, John Wiley, Chichester, UK, pp. 205-217.

Reid, W.V., 1998. Biodiversity hotspots. Trends in Ecology and Evolution 13, 275-280.

Rey Benayas, J..M., de la Montaña, E., 2003. Identifying high-value vertebrate diversity areas for strengthening nature conservation. Biological Conservation 114, 357-370.

Rey Benayas, J.M., Scheiner, S.M., 2002. Plant diversity, biogeography, and environment in Iberia: patterns and inferred mechanisms. Journal of Vegetation Science 13, 245-258.

Rey Benayas, J.M., Scheiner, S.M., García Sánchez-Colomer, M., Levassor, C., 1999. Commonness and rarity: theory and application of a new model to Mediterranean montane grasslands. Conservation Ecology 3 (5) [online] URL: http://www.consecol.org/vol3/iss1/art5.

Reyers, B., van Jaarsveld, A.S., Kruger, M., 2000. Complementarity as a biodiversity indicator strategy. Proceedings of the Royal Society of Biological Science B 267, 505-513.

Richter, B.D., Braun, D.P., Mendelson, M.A., Master, L.L., 1997. Threats to imperiled freshwater fauna. Conservation Biology 11, 1081-1093.

Ricketts, T.H., Dinerstein, E., Olson, D.M., Loucks, C.J., Eichbaum, W., DellaSala, D., Kavanagh, K., Hedao, P., Hurley, P.T., Carney, K.M., Abell, R., Walters, S., 1999. Terrestrial Ecoregions of North America: a Conservation Assessment. Island Press, Washington, USA.

Rodrigues, A.S., Andelman, S.J., Bakarr, M.I., Boitani, L., Brooks, T.M., Cowling, R.M., Fishpool, L.D., Da Fonseca, G.A., Gaston, K.J., Hoffmann, M., Long, J.S., Marquet, P.A., Pilgrim, J.D., Pressey, R.L., Schipper, J., Sechrest,W., Stuart, S.N., Underhill, L.G., Waller, R.W., Watts, M.E., Yan, X.,2004. Effectiveness of the global protected area network in representing species diversity. Nature 428(6983), 640-643.

Rossel, C., 2003. Recuperación de hábitats para la fauna y prevención de impactos. In: Rey Benayas, J. M., Espigares, T. and Nicolau, J.M. (Eds.). Restauración de ecosistemas mediterráneos. Servicio de Publicaciones de la Universidad de Alcalá, Alcalá de Henares, Spain.

Rossi, E., Kuitunen, M., 1996. Ranking of habitats for the assessment of ecological impact in land use planning. Biological Conservation 77, 227-234.

Rouse, J.D., Bishop, C.A., Struger, J., 1999. Nitrogen pollution: An assessment of its threat to amphibian survival. Environmental Health Perspectives 107, 799-803.

Sarkar, S., Margules, C., 2002. Operationalizing biodiversity for conservation planning. Journal of Bioscience 27, 299-308.

Scott, J.M., Davis, F.W., McGhie, R.G., Wright, R.G., Groves, C., Estes, J., 2001. Nature reserves: do they capture the full range of America´s biological diversity? Ecological Applications 11, 999-1007.

Semlitsch, R.D., 2000. Principles for management of aquatic-breeding amphibians. Journal of Wildlife Management 64, 615-631.

Semlitsch, R.D., Bodie, J.R., 2003. Biological criteria for buffer zones around wetlands and riparian habitats for amphibians and reptiles. Conservation Biology 17 (5), 1219-1228.

Stow, A.J., Sunnucks, P., Briscoe, D.A., Gardner, M.G., 2001. The impact of habitat fragmentation on dispersal of Cunningham's skink (*Egernia cunninghami*): evidence from allelic and genotypic analyses of microsatellites. Molecular ecology 10(4), 867-878.

Terborgh, J., 1999. Requiem for Nature. Island Press, Washington, USA.

Tilman, D., 1999. The ecological consequences of changes in biodiversity: a search for general principles. Ecology 80, 1455-1474.

Treweek, J.R., Hankard, P., Roy, D.B., Thompson, S., 1998. Scope for strategic ecological assessment of trunk-road development in England with respect to potential impacts on lowland heathland, the Darford warbler (*Sylvia undata*) and the sand lizard (*Lacerta agilis*). Journal of Environmental Management 53, 147-163.

Usher, M.B., 1986. Wildlife Conservation Evaluation. Chapman and Hall, London, UK.

Vargas, J.M., Real, R., 1997. Biogeografía de los anfibios y reptiles de la península ibérica. In: Pleguezuelos, J.M. (Ed.), Distribución y biogeografía de los anfibios y reptiles en España y Portugal, Monografías de Herpetología 3, Universidad de Granada, Asociación Herpetológica Española, pp. 309-320.

Vargas, J.M., Real, R., Guerrero, J.C., 1998. Biogeographical regions of the Iberian Peninsula based on freshwater fish and amphibian distributions. Ecography 21, 371-382.

Virolainen, K.M., Nättinen, K., Suhonen, J., Kuitunen, M., 2001. Selecting herb-rich forest networks to protect different measures of biodiversity. Ecological Applications 11, 411-420.

Williams, P.H., Araujo, M.B., 2000. Using probability of persistence to identify important areas for biodiversity conservation. Proceedings of the Royal Society London B 267, 1959-1966.

Williams, P.H., Humphries, C.J., Vane-Wright, R.L., 1991. Measuring biodiversity: taxonomic relatedness for conservation priorities. Australian Systematic Botany 4, 665-679.

Williams, P., Gibbons, D., Margules, C., Rebelo, A., Humphries, C., Pressey, R., 1996. A comparison of richness hotspots, rarity hotspots and complementarity areas for conserving diversity using British birds. Conservation Biology 10, 155-174.

Williams, P.H., Margules, C.R., Hilbert, D.W., 2002. Data requirements and data sources for biodiversity priority area selection. Journal of Bioscience 27, 327-338.

Willson, J.D., Dorcas, M. E., 2003. Effects of habitat disturbance on stream salamanders: Implications for buffer zones and watershed management. Conservation Biology 17 (3), 763-771.

Woodford, J.E., Meyer, M.W., 2003. Impact of lakeshore development on green frog abundance. Biological Conservation 110 (2), 277-284.

Appendix 3.A. List of species used in the identification of areas of high diversity. The original raw data were extracted from Pleguezuelos (1997). The number in parenthesis refers to the number of 20 x 20 km cells where the species was present. The vulnerability status according to the Red Book of Spanish Vertebrates (Blanco & González 1992) is indicated. Categories are defined as follows. Endangered: species in danger of extinction and unlikely to survive if causal factors persist; Vulnerable: species that would soon belong to the endangered category if causal factors persist; Rare: species with small populations at risk mostly because they extend on small geographical ranges or habitats or because their populations are sparse; Undetermined: species that do belong to the endangered, vulnerable or rare categories, but the current knowledge does not allow a certain assignment; Insufficiently known: species that are suspected to belong to the former categories, but there is no certainty about that; Non-threatened: species with no evident threats. Abbreviations: E = endangered, V = vulnerable, R = rare, U = undetermined, I-K = insufficiently known, I = introduced, and N-T = non-threatened.

Amphibians

Alytes cisternasii (139) (N-T), *Alytes dickhilleni* (43), *Alytes muletensis* (2) (E), *Alytes obstetricans* (409) (N-T), *Bufo bufo* (786) (N-T), *Bufo calamita* (617) (N-T), *Bufo mauritanicus* (1), *Bufo viridis* (21) (R), *Chioglossa lusitanica* (48) (R), *Discoglossus galganoi* (331) (N-T), *Discoglossus pictus* (11) (I), *Euproctus asper* (69) (N-T), *Hyla arborea* (216) (N-T), *Hyla meridionalis* (150) (N-T), *Pelobates cultripes* (430) (N-T), *Pelodytes punctatus* (303) (N-T), *Pleurodeles waltl* (270) (N-T), *Rana catesbeiana* (1), *Rana dalmatina* (12) (V), *Rana iberica* (142) (N-T), *Rana perezi* (881) (N-T), *Rana pyrenaica* (5), *Rana temporaria* (161) (N-T), *Salamandra salamandra* (376) (N-T), *Triturus alpestris* (41) (R), *Triturus boscai* (237) (N-T), *Triturus helveticus* (172) (N-T), and *Triturus marmoratus* (364) (N-T).

Note: *A. dickhilleni* and *R. pyrenaica* have recently been catalogued; *B. mauritanicus* and *R. cartesbeiana* are introduced species and are not included in the Red Book.

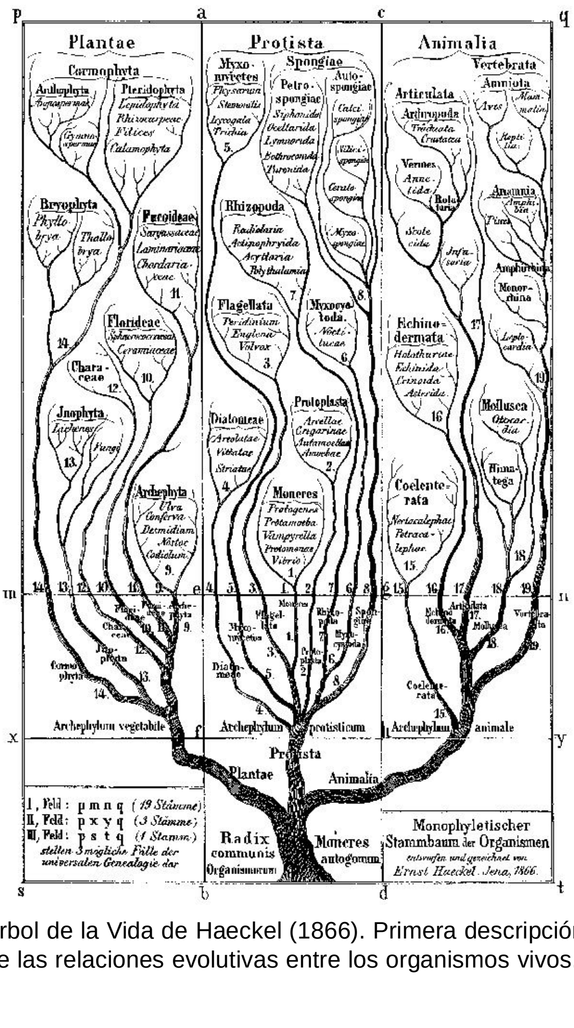
Reptiles

Acanthodactylus erythrurus (187) (N-T), *Algyroides marchi* (11) (R), *Anguis fragilis* (301) (N-T), *Anolis carolinensis* (1), *Blanus cinereus* (238) (N-T), *Coluber hippocrepis* (305) (N-T), *Coluber viridiflavus* (46) (R), *Coronella austriaca* (222) (N-T), *Coronella girondica* (495) (N-T), *Chalcides bedriagai* (199) (N-T), *Chalcides striatus* (402) (N-T), *Chamaeleo chamaeleon* (30) (E), *Elaphe longissima* (58) (R), *Elaphe scalaris* (664) (N-T), *Emys orbicularis* (109) (V), *Hemidactylus turcicus* (152) (N-T), *Lacerta agilis* (4) (V), *Lacerta araica* (2) (E), *Lacerta aurelioi* (3) (E), *Lacerta bonnali* (7) (U), *Lacerta lepida* (904) (N-T), *Lacerta monticola* (55) (N-T), *Lacerta perspicillata* (5) (I), *Lacerta schreiberi* (177) (N-T), *Lacerta viridis* (129) (N-T), *Lacerta vivipara* (77) (N-T), *Macroprotodon cucullatus* (172) (N-T), *Malpolon monspessulanus* (752) (N-T), *Mauremys leprosa* (257) (N-T), *Natrix maura* (841) (N-T), *Natrix natrix* (462) (N-T), *Podarcis bocagei* (128) (N-T), *Podarcis hispanica* (752) (N-T), *Podarcis lilfordi* (12) (V), *Podarcis muralis* (192) (N-T), *Podarcis pityusensis* (12) (R), *Podarcis sicula* (8) (I), *Psammodromus algirus* (771) (N-T), *Psammodromus hispanicus* (389) (N-T), *Pseudemys picta* (1), *Tarentola mauritanica* (389) (N-T), *Testudo graeca* (21) (E), *Testudo hermanni* (34) (V), *Trachemys scripta* (46), *Trionyx spiniferus* (5), *Vipera aspis* (129) (N-T), *Vipera latastei* (263) (N-T), and *Vipera seoanei* (136) (N-T).

Note: *A. carolinensis*, *P. picta*, *T. scripta* and *T. spiniferus* are introduced species that are not included in the Red Book.

Capítulo 4

"...Con la variedad se adorna la Naturaleza"
Vicente Espinel



Árbol de la Vida de Haeckel (1866). Primera descripción de las relaciones evolutivas entre los organismos vivos.

Capítulo 4

Planificación sistemática para la conservación de la diversidad de vertebrados. Un caso de estudio en una región europea mediterránea.

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

Enrique de la Montaña, José M. Rey Benayas, Irene Razola, Ana Vasques. En revisión. Systematic conservation planning of vertebrate diversity. A case study in a Mediterranean European region. Enviado a Conservation Biology.

Resumen

La preservación de la biodiversidad necesita aproximaciones sistemáticas para la planificación de la conservación. Las redes de áreas protegidas son herramientas importantes para alcanzar los objetivos de conservación. Para mejorar la efectividad de las áreas protegidas, la selección de áreas prioritarias debería incluir consideraciones de tres componentes fundamentales: elementos especiales de conservación, especies clave y representación. Presentamos una aproximación con tres enfoques relativos a estos tres componentes para la planificación de la conservación de vertebrados en Castilla-La Mancha, España. Como elementos especiales de conservación, identificamos Áreas Prioritarias para la Conservación de especies utilizando cuatro criterios: riqueza de especies, rareza geográfica, vulnerabilidad de especies y un índice combinado de esos tres criterios. La Red Natura 2000 se usó para incluir en nuestra aproximación áreas para las especies clave. Evaluamos la representación en la Red Natura 2000 de todos los tipos de uso del suelo y realizamos un análisis de huecos con las Áreas Prioritarias para la Conservación. Finalmente, combinamos esas Áreas Prioritarias para la Conservación, las áreas de conservación existentes y áreas de conectividad, mediante un análisis de coste-distancia, para definir la configuración espacial para la conservación de los vertebrados. El Índice Combinado de Biodiversidad fue el criterio analizado más eficiente para identificar las Áreas Prioritarias para la Conservación. La Red Natura 2000 mostró un alto porcentaje de coincidencia con las Áreas Prioritarias para la Conservación, mientras que la Red de Espacios Naturales Protegidos mostró una baja coincidencia. Aunque la Red Natura 2000 incrementó la superficie protegida en la región, seis habitat agrícolas no fueron representados adecuadamente. Encontramos una correlación positiva moderada entre la vulnerabilidad de aves y los tres principales agroecosistemas subrepresentados. Según nuestra aproximación de múltiples componentes, aproximadamente el 29% del área de estudio fue necesaria para capturar dos importantes elementos de la biodiversidad: el 100% de las especies de vertebrados y todos los tipos de paisaje. Nuestros resultados muestran que las redes de áreas de conservación existentes son insuficientes para representar con garantías de conservación la biodiversidad de la región de estudio. Para el fortalecimiento de la conservación de la biodiversidad son importantes áreas adicionales con relevantes características de diversidad, áreas de conectividad y el establecimiento de objetivos de conservación fuera de las áreas protegidas.

Palabras clave: Agroecosistemas; Análisis del camino de menor coste; Análisis de huecos; Conectividad; Índice combinado de biodiversidad; Rareza; Red Natura 2000; Representación de hábitats

Systematic conservation planning of vertebrate diversity. A case study in a Mediterranean European region

Enrique de la Montaña, José M. Rey-Benayas, Irene Razola, Ana Vasques

Departamento de Ecología, Universidad de Alcalá. Alcalá de Henares, 28871 Madrid, Spain.

Abstract

Preserving biodiversity needs systematic approaches to conservation planning. Protected area networks are an important tool to achieve conservation goals. To improve effectiveness of protected areas, selection of priority areas should include consideration of three main components: special conservation elements, focal species and representation. We present a three-track approach relative to these three components for vertebrate conservation planning in Castilla-La Mancha, Spain. As special conservation elements, we identified Priority Areas for Conservation of species using four criteria: species richness, geographic rarity, species vulnerability and a combined index of these three criteria. The Natura 2000 Network was used to include in our approach conservation areas for focal species. We evaluated the representation of every land-use type in the Natura 2000 Network and we performed a gap analysis with Priority Areas for Conservation. Finally, we combined these Priority Areas for Conservation, existing conservation areas and connectivity areas, by conducting a cost-distance analysis, to delineate the spatial configuration for vertebrate conservation. The Combined Index of biodiversity was the most efficient criterion analyzed to identify Priority Areas for Conservation. The Natura 2000 Network showed a high percentage of coincidence with identified Priority Areas for Conservation, whereas the Natural Protected Areas Network had a low percentage coincidence. Although the Natura 2000 Network increased the protected area in the region, six agricultural habitats were inadequately represented. We found moderate positive correlations between vulnerability of birds and the three main under-represented agroecosystems. According to our multi-track approach, ~29% of study area was required to capture two important elements of biodiversity: 100% of vertebrate species and all landscape types. Our results show that existing conservation area networks are insufficient to represent with guarantees of conservation the biodiversity of the study region. Additional areas with outstanding features of diversity, connectivity areas and establishment of targets for off-reserve conservation are of fundamental importance for strengthening biodiversity conservation.

Keywords: Agroecosystems; Connectivity; Combined Index of biodiversity; Gap analysis; Least-cost path analysis; Natura 2000 Network; Rarity; Habitat representation

Introduction

Systematic approaches to conservation planning have been developed over the last 20 years in order to achieve clearly stated conservation goals (Margules & Pressey 2000; Cowling & Pressey 2003). There is little doubt that establishing protected areas is an important tool for conservation (Soulé 1991), and constitutes the cornerstone on which local, regional and global strategies are built. However, the effectiveness of protected areas in representing biodiversity has been frequently questioned (Scott *et al*. 2001; Andelman & Willig 2003; Gaston *et al*. 2006), and it is accepted that existing conservation areas usually provide inadequate coverage to biodiversity (Rodrigues *et al*. 2004). The major cause is that economic and development interests are often opposed to conservation goals, but also because of the array of different reasons that motivate the establishment of protected areas. Thus, selection of critical areas for biodiversity conservation needs to set prior targets and precise prescriptions (Soulé & Sanjayan 1998; Myers *et al*. 2000; Pimm *et al*. 2001; Underwood *et al*. 2008). For conservation planning to be relevant, systematic approaches that integrate consideration of special conservation elements, focal species and representation are suggested (Noss *et al*. 1999).

However, to date, very few applications integrate multiple components into regional conservation plans (Hoctor *et al*. 2000; Noss *et al*. 2002; Cowling *et al*. 2003; Beazley *et al*. 2005). The European Union Natura 2000 Network (Directive 92/43/EEC) promotes the maintenance of biodiversity by means of protecting the distribution areas of focal species of wild fauna and flora (the so called "species of Community interest") and of the ecosystems that are their habitat. It also provides protection to natural habitats *per se* of Community interest because they (1) are in danger of disappearance; (2) have a small natural distribution area; and/or (3) present outstanding examples of typical characteristics of European biogeographical regions. However, in many parts of Europe, besides "natural habitats", there are agricultural landscapes that are over several centuries old (Williamson 1986; Groppali 1993). Many species that occur in these agricultural landscapes are not well protected (Seoane *et al*. 2006). Accordingly, effective conservation planning should take into consideration habitats that include every type of landscape.

In this study, we used a three-track approach to vertebrate conservation planning that integrated special elements, focal species and representation. We defined two conservation targets: (1) to obtain protection for 100% of species in a region; and (2) representation of every type of landscape. We applied this approach to a case study in central Spain as an illustrative example. We identified priority areas for conservation (PACs) as special conservation elements. These areas have high conservation value for several biodiversity components (species richness, species rarity and threatened species) (Rey Benayas & de la Montaña 2003) and fulfill one of the major objectives for the establishment of conservation areas; i.e., to maximise the number of species conserved with the minimum land required (Cabeza & Moilanen 2001). To address the issue of representation, we used a land-cover database to evaluate existing conservation areas, and ensured that every important landscape for the maintenance of biodiversity in this humanised area was represented.

We also evaluated the efficiency of the various criteria used to identify the PACs and the correlation of these criteria with the land-use types. Next, we performed a gap analysis by overlaying existing conservation areas with the locations of the identified PACs, and evaluated the representation of every landscape. Finally, we delineated the spatial configuration for vertebrate conservation by combining the identified PACs, existing conservation areas and connectivity areas on the basis of least-cost paths. Our analyses are illustrative, not exhaustive, but they provide an example of systematic planning for conservation of biodiversity. A similar approach can be used elsewhere using different species groups, criteria, threats or scales.

Material and methods

Study area

Castilla-La Mancha is a region located in central Spain (Fig. 4.1). It is 79222 km^2 in extent. It is surrounded on all sides by mountains; two additional mountain systems together with the vast southern spanish plateau complete the relevant geomorphologic units. The area is mostly devoted to agricultural activities. Climate is continental Mediterranean, with dry, hot summers and cold winters. There is a variety of climatic areas, mostly related to altitude differences. This causes considerable variation of vegetation composition and structure.

Criteria for identifying Priority Areas for Conservation

We used four criteria to identify PACs for vertebrate species: species richness, rarity, vulnerability, and a combined index of these three criteria. The sources of the species distribution data were national atlases (Ministerio de Medio Ambiente 2002a, b, 2003). These atlases provided information on species distribution based on their presence in 10 ×10 km cells, with a total of 906 cells in the study region.

Rarity of a species *i* was defined by its geographical range measured as the inverse of the number of cells in which it was present ($1/n_i$). For a cell *r*, the rarity index was $\sum_{i=1}^{S}(1/n_{ri})/s_r$, where s_r was the number of species found in the cell.

Vulnerability is a surrogate concept of rarity plus rates of habitat loss and other threats. Species vulnerability was quantified using the categories defined by the International Union for Conservation of Nature (IUCN 2001). The following species categories were considered: critically endangered, endangered, vulnerable, near threatened and least concern. We assigned every category a score related to its degree of vulnerability: 5 for critically endangered species, 4 for endangered species, 3 for vulnerable species, 2 for near threatened species, and 1 for species of least concern. We acknowledge the subjectivity of these scores; they merely represent a rank and have a relative value, and any other choice would have been equally subjective. For a cell, the vulnerability index was $\sum_{i=1}^{S}V_{ri}/s_r$, where V_{ri} was the vulnerability score of the species *i* present in the cell. We used the Combined Index of species richness, rarity and vulnerability defined by Rey Benayas and de la Montaña (2003): $\sum_{i=1}^{S}(1/n_{ri})V_{ri}$. In this index, species richness is implicit in $\sum_{i=1}^{S}$.

We also used a Standardized Biodiversity Index (SBI) that measured species richness, rarity and vulnerability of all four *taxa* together in every cell. We standardized by dividing the combined index of biodiversity of each taxonomic group in every cell by its mean across all cells, and then added up the four standardized combined indices. The Standardized Biodiversity Index formula is:

$$\sum_{j=1}^{4} 1/m_j \sum_{i=1}^{jS} (1/n_{ji})V_{ji}$$

where m_j refers to the mean combined index of biodiversity of the taxonomic group j across the cells.

Next, all diversity indices for the *taxa* were ranked. To quantitatively define PACs, we considered the pool of cells within the upper ranked values for the various criteria that included all species. We also determined the number of cells necessary to capture all threatened species.

Existing conservation areas in the region

There are 30 main protected areas (two national parks, six natural parks and 22 natural reserves) in the region that have a protection level according to IUCN categories II, IV and V (IUCN 1994), and represent 3.5% of the total territorial area (Fig. 4.1). Conservation goals of the European Union have motivated the development of the Natura 2000 Network in the last decade. When this network has been completed, the sites of Community importance determined by the Habitats Directive (92/43/EEC) and the areas established by the Birds

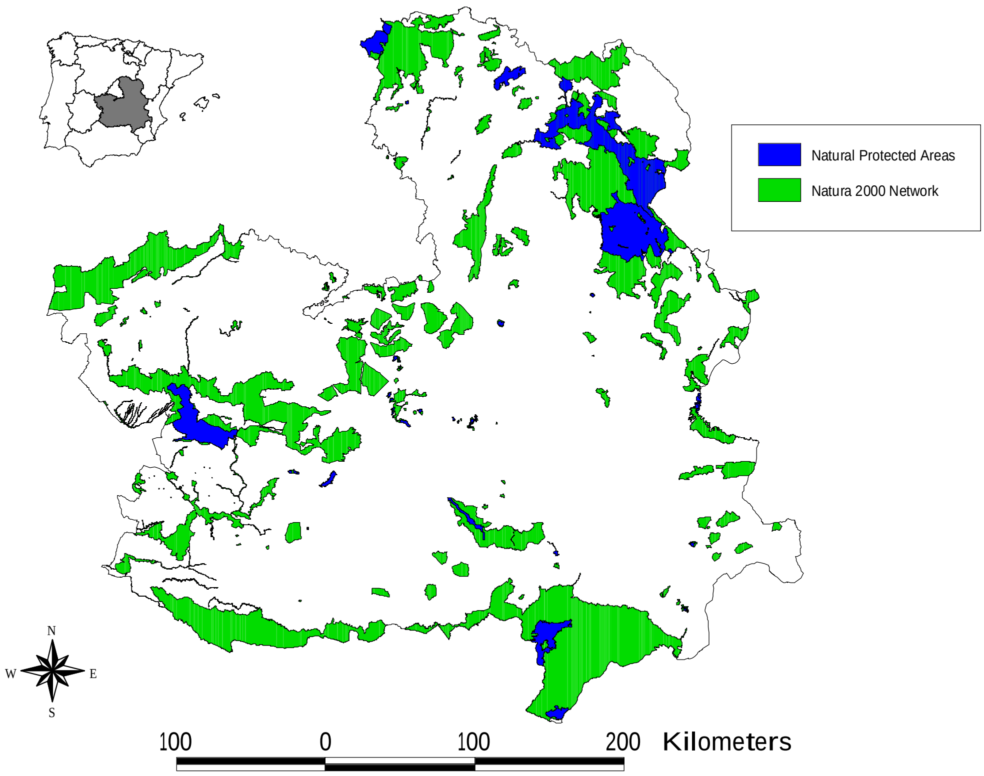


Figure 4.1. Map of Castilla-La Mancha in central Spain with the Natural Protected Areas and sites of Community importance established by the Natura 2000 Network.

Directive (79/409/EEC) will represent 22.9% of the study area. The current Natural Protected Areas in the study region have been proposed as areas of Community importance, and therefore, will be included in the Natura 2000 Network. Gap analysis between PACs and conservation areas was based on χ^2 test.

We used the CORINE Land Cover (European Commission 1993) to evaluate the representation of all habitat types in the existing conservation areas network, regardless of their anthropogenic origin and maintenance. To simplify the analysis, the initial 85 categories of land use were reclassified into 28 broader categories (Fig. 4.2). The resulting land-use map was overlapped with the Natura 2000 Network. We deemed a habitat type as under-represented if less than 15% of its total area was included in the Natura 2000 Network (Fig. 4.2). We chose this threshold arbitrarily because there are no standard guidelines that refer to the percentage area of each habitat type that should be included in a conservation plan, and because the commonly used 10 or 12% is considered insufficient to achieve conservation goals (Soulé & Sanjayan 1998; Margules & Pressey 2000).

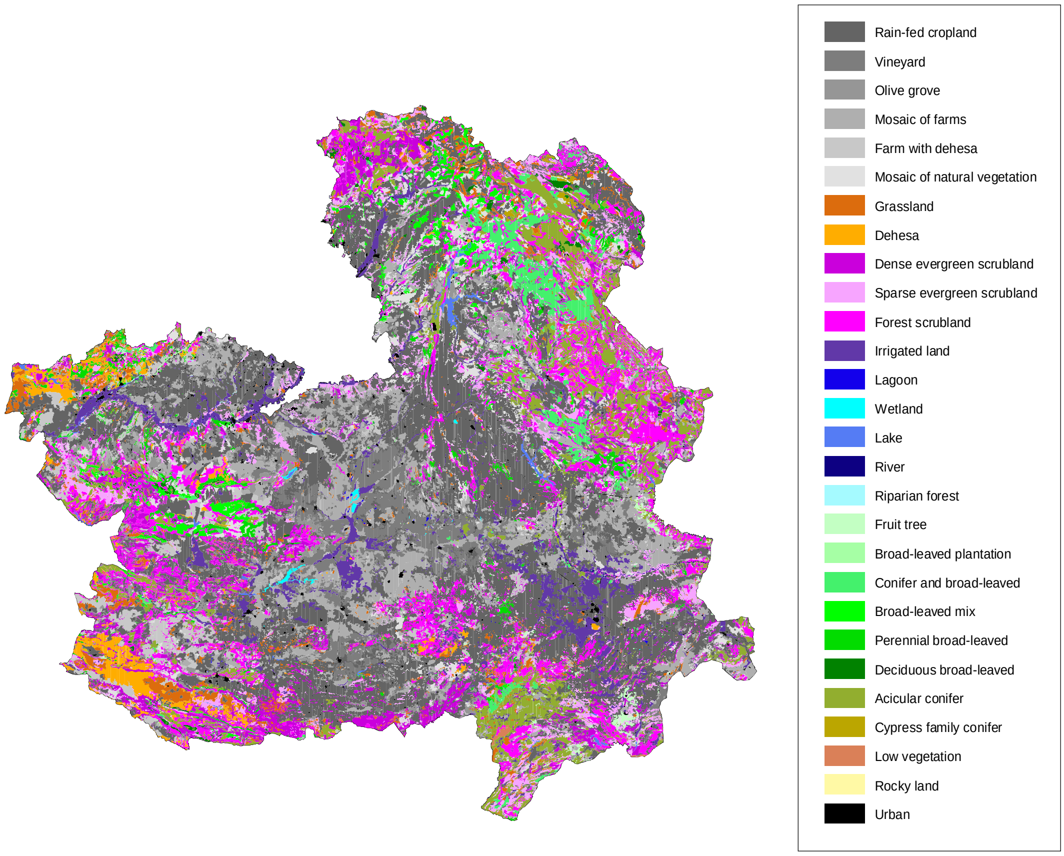


Figure 4.2. There were 28 new categories in this land use classification reclassified from the initial 85 categories considered by CORINE Land Cover database in Castilla-La Mancha. Categories considered under-represented by existing protection areas are showed in shades of grey (see Table 4.3).

We used a correlation analysis and Bonferroni corrections for multiple comparisons to examine the relationships between land-use types and the criteria used to identify PACs.

Selecting areas for conservation planning

To include unprotected areas that were detected by gap analysis, we combined the identified PACs with the Natura 2000 Network. These two figures represent special conservation elements and focal species because they provide areas with high biodiversity value and habitats for species of Community interest. However, they may still inadequately represent all the important habitat types for biodiversity preservation in the region. Thus, we selected additional areas of under-represented habitat types, and gave priority to patches that improved connectivity between the largest areas delineated by merged PACs and the Natura 2000 Network. In order to provide supplementary habitats for focal species and opportunities for dispersal.

We selected connectivity areas by conducting cost-distance analysis among target areas that contained under-represented habitats. The least-cost path resulting represents the least amount of resistance for species movement between habitats and is a function of distance, habitat suitability and obstacles. We created cost-surface maps by combining habitat suitability and planned infrastructures for the next few years (planned highways and roads, high-speed railway lines, gas pipelines, one airport, one theme park, wind farms and water reservoirs and pipelines), in order to avoid future impacts. We also considered for the cost-surface maps zones of high wildlife mortality ("black spots") established by environmental organisations. In particular, black spots for birds are areas in which there are a high number of electrocutions and collisions with power lines, whereas other wildlife black spots refer to areas with high number of road kills. Cost-distance analyses were completed in ArcView 3.2 (ESRI 1999). First, we used the nearest features extension (Jenness 2007) to select the largest (>10000 ha) and nearest target areas. Secondly, we used the pathmatrix extension (Ray 2005) to find least-cost paths across cost-surface grids, and then manually selected connectivity areas to achieve at least 15% representation of each under-represented habitat. Finally, we overlaid the new connectivity areas selected with PACs and the Natura 2000 Network to create a synthesis layer.

Results

Distribution and evaluation of Priority Areas for Conservation

For the four *taxa*, the mean percentage of cells that was necessary to retain all species was 2.8% for the Combined Index, 4.4% for rarity, 21.5% for richness and 46.7% for vulnerability. For threatened species, it was 2.4%, 4.3%, 21.4% and 46.7%, respectively (Table 4.1). Thus, evaluation of PACs demonstrated that the Combined Index of biodiversity was the most efficient criterion to identify areas for protection of vertebrate diversity in Castilla-La Mancha, since it required the lowest number of cells to retain 100% of all species and of all threatened species of reptiles, breeding birds and mammals. The rarity index required the same number of cells as the Combined Index to retain all mammal species and was the most efficient criterion for all species and threatened species of amphibians.

Table 4.1. Number (and proportion in parenthesis) of cells that were required to retain all species and all threatened species of amphibians, reptiles, breeding birds, and mammals according to the different criteria used to identify Priority Areas for Conservation.

	Amphibians		Reptiles		Breeding birds		Mammals	
	All species	Threatened species	All species	Threatened species	All species	Threatened species	All species	Threatened species
Richness	33 (3.6%)	33(3.6%)	12 (1.3%)	8 (0.9%)	487 (53.8%)	487 (53.8%)	248 (27.4%)	248 (27.4 %)
Rarity	19 (2.1%)	19(2.1%)	57 (6.3%)	57 (6.3%)	66 (7.3%)	66 (7.3%)	16 (1.8%)	13 (1.4%)
Vulnerability	76 (8.4%)	76 (8.4%)	234 (25.8%)	196 (21.6%)	712 (78.6%)	375 (41.4%)	670 (74.0%)	647 (71.5%)
Combined Index	23 (2.5%)	23 (2.5%)	12 (1.3%)	7 (0.8%)	52 (5.7%)	52 (5.7%)	16 (1.8%)	6 (0.7%)

Table 4.2. Gaps (%) between Priority Areas for Conservation defined by (i) the Combined Index of each taxonomic group and (ii) by the Standardized Biodiversity Index (SBI), with the existing protected area networks.

	Amphibians	Reptiles	Breeding birds	Mammals	SBI
Natura 2000 Network	21.7	0	17.3	6.3	11.5*
Natural protected areas	82.6	58.3	69.2	68.8	72.7*

*P< 0.05

One hundred and twenty-one cells (13.3% of the total) highlighted by the Standardized Biodiversity Index of all *taxa* were necessary to retain 100% of species (Fig. 4.3). There was an aggregation of PACs at the southern and northern peripheral mountains, whereas they were distributed sparsely in the central part of the region.

Coincidence of Priority Areas for Conservation and existing conservation areas

There was a low percentage of gaps between the Natura 2000 Network and PACs, based on the Combined Index of the different *taxa* (<22%, mean 11.3%). In contrast, there was a high percentage of gaps between the Natural Protected Areas and PACs (>58%, mean 69.7%) (Table 4.2). The gaps between PACs according to the Combined Index and both conservation networks followed the order amphibians > breeding birds > mammals > reptiles. Percentages for the Standardized Biodiversity Index were close to the reported means, with 11.5% of gaps for the Natura 2000 Network (χ^2 = 55.20, P <0.000) and 72.7% of gaps for the Natural Protected Area Network (χ^2 = 10.38, P <0.015). Additionally, there were 9.5% and 33.3% of cells identified as PACs with <10% of their area included in the Natura 2000 and Natural Protected Areas Networks, respectively.

Habitat representation in the Natura 2000 Network

We found that eight out of the 28 classes were inadequately represented (<15% of their area) by the Natura 2000 Network (Table 4.3). Two of these classes were urban land and irrigated land, which are of little importance for the maintenance of biodiversity in the study area, thus we do not consider urban and irrigated land in the analysis. The other land-use types under-represented were agricultural habitats. Vineyard (4%), olive grove (6.5%) and rain-fed cropland (10.3%) are traditional Mediterranean

Table 4.3. Total area of each land-use type in Castilla-La Mancha; area and percentage included in Natura 2000 Network; and increase if Priority Areas for Conservation (PACs) defined by the Standardized Biodiversity Index are added to Natura 2000 Network.

Land-use	Total area (ha)	Area in Natura (ha)	% in Natura	% in Natura-PACs	% increase
Lagoon	4285	3209	74.9	77.1	3.0
Rocky land	1564	1109	70.9	70.9	0.0
Cypress family conifer	19909	13804	69.3	70.7	2.0
Deciduous broad-leaved	47625	31413	66.0	67.5	2.3
Conifer and broad-leaved	181505	113592	62.6	63.3	1.2
Acicular conifer	566663	313943	55.4	57.7	4.2
Wetland	9149	5031	55.0	72.2	31.4
Low vegetation	16896	8917	52.8	60.5	14.6
Broad-leaved mix	119039	56520	47.5	49.7	4.6
Dense evergreen shrubland	452621	189285	41.8	50.4	20.5
Perennial broad-leaved	149908	61523	41.0	43.8	6.7
Dehesa	134912	52463	38.9	57.9	48.9
Broad-leaved plantation	6441	2450	38.0	49.9	31.2
Forest shrubland	819177	304910	37.2	41.4	11.1
Fruit tree	23098	8161	35.3	36.0	2.0
River	11362	3589	31.6	32.9	4.2
Sparse evergreen shrubland	435695	134901	31.0	37.0	19.4
Lake	33533	9171	27.3	31.7	16.0
Riparian forest	2978	756	25.4	25.7	1.4
Grassland	299532	72679	24.3	33.2	36.7
Mosaic of natural vegetation	292354	43497	14.9	18.0	21.2
Farm with *dehesa*	215155	29487	13.7	24.2	76.6
Rain-fed cropland	2288431	235351	10.3	14.5	41.3
Irrigated land	371555	30672	8.3	13.2	60.0
Mosaic of farms	796706	55565	7.0	11.1	58.8
Olive grove	193265	12566	6.5	11.6	78.1
Vineyard	369403	14695	4.0	11.6	192.6
Urban	77644	3037	3.9	5.7	46.1

farm systems that extend over large areas. However, their individual patches are frequently of little surface and are found in combination with themselves and with other types of natural vegetation, which gives rise to agroecosystems. Mosaic of farms (7%), farm with *dehesa* (13.7%) or mosaic of natural vegetation (14.9%) were also inadequately represented. These six under-represented habitats were 41550 km^2 in extent, or 52.3% of the study area. Lagoons were the habitat best represented in the Network (~75% of area).

PACs significantly improved habitat representation, with a mean increase of ~76% in under-represented land-use types (Table 4.3). There was also a high increase in the representation of important habitats for biodiversity conservation in humanised landscapes: *dehesas*, grasslands and wetlands (~49%, ~37% and ~31%, respectively).

Association between land-use types and diversity features

After applying corrections for multiple comparisons, correlation coefficients between land-use types and criteria used to identify PACs showed that forest ecosystems, such as deciduous broad-leaved, acicular conifers and dense evergreen shrubland, were positively correlated with scores for amphibian and mammal diversity, chiefly (Table 4.4). Dense evergreen shrubland was also positively correlated with the Standardized Biodiversity Index of all *taxa*. However, these forest ecosystem types were only negatively correlated with vulnerability of breeding birds, as sparse evergreen shrubland, forest shrubland, perennial broad-leaved and mix of conifer and broad-leaved species.

These results were the opposite in the agroecosystems: rain-fed cropland, vineyard, mosaic of farms, irrigated land and mosaic of natural vegetation, in which most significant

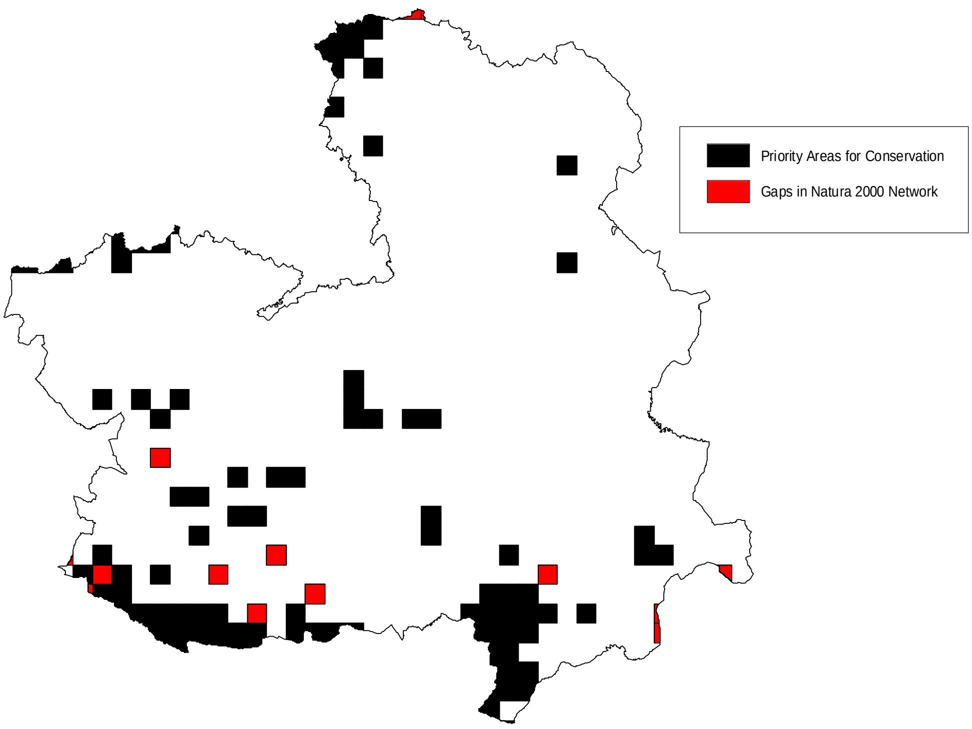


Figure 4.3. Distribution of 121 Priority Areas for Conservation in the 10 × 10 km study area that includes 100% of vertebrate species. Fourteen of these Priority Areas for Conservation (in red) are not currently included within existing protected areas.

Table 4.4. Correlation coefficients between land-use types and criteria used to identify Priority Areas for Conservation: species richness (RICH), species rarity (RAR), species vulnerability (VULN), Combined Index (CI) and Standardized Biodiversity Index (SBI).

Land-use	Amphibians				Reptiles				Breeding birds				Mammals				SBI
	RICH	RAR	VULN	CI	RICH	RAR	VULN	CI	RICH	RAR	VULN	CI	RICH	RAR	VULN	CI	
Rain-fed cropland	-0.35*	-0.27*	-0.28*	-0.29*	-0.24*	-0.09	-0.18*	-0.18*	-0.26*	-0.09	0.36*	-0.03	-0.10	-0.22*	0.11	-0.21*	-0.32*
Vineyard	-0.18*	-0.14*	-0.20*	-0.12	-0.07	-0.01	-0.07	-0.08	-0.15*	0.12	0.33*	0.15*	-0.12	-0.12	0.14*	-0.11	-0.07
Irrigated land	-0.18	-0.11	-0.10	-0.12	-0.07	-0.04	-0.08	-0.06	-0.11	0.08	0.23*	0.08	0.02	-0.10	0.12	-0.06	-0.08
Fruit tree	-0.07	-0.01	0.01	-0.03	-0.01	-0.01	-0.02	-0.02	-0.03	-0.03	-0.07	-0.03	0.17*	0.01	-0.01	0.05	-0.01
Olive grove	0.05	-0.02	0.04	-0.04	0.02	-0.02	0.04	0.01	-0.05	-0.01	0.01	-0.01	-0.14*	-0.06	0.01	-0.10	-0.06
Mosaic of natural	-0.30*	-0.19*	-0.24*	-0.20*	-0.09	-0.02	-0.06	-0.09	-0.24*	0.01	0.33*	0.02	-0.17*	-0.21*	0.13*	-0.18*	-0.20*
Mosaic of farms	-0.15*	-0.08	-0.07	-0.14	-0.09	-0.04	-0.06	-0.06	-0.02	-0.11	-0.18*	-0.09	-0.16*	-0.03	-0.16*	-0.11	-0.17*
Grassland	0.22*	0.08	0.10	0.10	0.02	-0.01	0.02	-0.01	0.12	0.01	-0.08	-0.07	-0.01	0.07	-0.09	0.02	0.06
Farm with *dehesa*	0.19*	0.01	0.10	0.05	0.01	0.03	-0.03	-0.01	0.01	0.06	0.08	0.07	-0.06	-0.02	0.01	-0.06	0.02
Dehesa	0.23*	0.11	0.11	0.18*	0.01	0.01	0.01	0.01	0.09	0.03	-0.01	0.02	-0.01	0.09	-0.01	0.04	0.10
Perennial broad-leaved	0.09	0.01	0.07	0.01	0.01	-0.01	0.01	-0.01	-0.01	-0.05	-0.14*	-0.05	-0.12	-0.03	-0.01	-0.08	-0.04
Deciduous broad-leaved	0.08	0.14*	0.02	0.04	0.06	0.03	0.01	0.09	0.02	0.01	-0.15*	-0.01	0.01	0.18*	-0.12	0.13*	0.07
Broad-leaved plantation	0.12	0.03	0.05	0.09	0.01	0.04	-0.03	0.09	0.04	0.01	-0.06	-0.01	0.05	0.02	-0.05	0.01	0.04
Broad-leaved mix	0.10	-0.01	0.04	-0.01	0.01	-0.01	0.03	-0.01	0.07	-0.04	-0.09	-0.02	-0.13*	0.01	-0.16*	-0.06	-0.03
Acicular conifer	0.04	0.11	0.13*	0.05	0.13*	0.01	0.08	0.09	0.03	-0.02	-0.38*	-0.06	0.24*	0.17*	-0.05	0.25*	0.12
Cypress family conifer	-0.10	-0.06	-0.13*	-0.05	-0.02	0.04	-0.04	-0.01	-0.01	-0.01	-0.09	-0.02	-0.02	0.03	-0.09	-0.01	-0.05
Conifer and broad-leaved	-0.04	-0.03	-0.01	-0.05	0.04	-0.02	0.01	0.02	0.03	-0.01	-0.23*	-0.03	-0.04	0.01	-0.15*	-0.01	-0.04
Dense evergreen shrubland	0.23*	0.17*	0.13*	0.10	0.10	0.07	0.06	0.09	0.15*	0.01	-0.21*	-0.02	0.08	0.23*	0.03	0.18*	0.14*
Sparse evergreen shrubland	-0.06	0.02	-0.01	-0.02	0.03	0.01	0.01	-0.01	0.07	-0.04	-0.15*	-0.04	0.09	0.09	-0.03	0.11	-0.02
Forest shrubland	0.08	0.05	0.11	-0.03	0.10	0.03	0.06	0.07	0.02	-0.06	-0.29*	-0.07	0.01	0.05	0.03	0.03	-0.02
Riparian forest	-0.01	-0.03	-0.05	-0.03	-0.08	-0.01	-0.05	-0.05	0.03	-0.01	-0.04	-0.01	-0.05	-0.02	-0.05	-0.03	-0.04
Rocky land	0.10	0.01	0.07	0.03	0.06	0.01	0.06	0.01	-0.01	-0.02	-0.06	-0.01	-0.05	-0.01	-0.02	-0.02	0.01
Low vegetation	0.12	0.14*	0.10	0.11	0.14*	0.03	0.09	0.12	-0.01	-0.01	-0.09	-0.01	0.07	0.09	-0.02	0.09	0.14*
Wetland	0.08	0.01	0.02	0.01	-0.01	-0.01	0.01	-0.01	0.12	0.61*	0.18*	0.72*	-0.05	-0.02	0.06	-0.03	0.23*
Lagoon	0.07	0.01	0.08	0.01	0.01	-0.02	0.01	-0.01	0.15*	0.43*	0.20*	0.49*	0.01	-0.02	0.09	-0.01	0.15*
River	-0.02	-0.03	0.02	-0.05	0.01	-0.02	-0.01	-0.02	0.09	0.02	-0.01	0.01	0.03	-0.06	0.14*	-0.02	-0.03
Lake	-0.02	-0.02	-0.02	-0.01	0.03	-0.02	0.03	0.01	0.05	0.07	-0.05	0.03	-0.01	-0.03	-0.01	-0.01	-0.02
Urban	-0.06	-0.07	-0.07	-0.08	-0.05	-0.05	-0.07	-0.05	-0.03	0.01	0.12	-0.01	-0.01	-0.05	0.05	-0.04	-0.08

*Coefficients that are significant at $P < 0.05$ after applying Bonferroni's corrections for multiple comparisons.

correlations were negative with several diversity scores of amphibians, reptiles and mammals, and with richness of breeding birds. The amounts of these agroecosystems (except of mosaic of natural vegetation) were, however, positively correlated with vulnerability of breeding birds. The amount of mosaic of natural vegetation, rain-fed cropland and mosaic of farms was also negatively correlated with SBI.

Otherwise, there was a positive correlation between diversity scores of richness of amphibians with *dehesa*, grassland and farm with *dehesa*, as well as Combined Index of amphibians with *dehesa*.

The amount of wetland and lagoon were positively correlated with SBI and with all diversity scores of breeding birds (except richness in wetlands). Also, low vegetation areas were positively correlated with SBI, as well as with rarity of amphibians and richness of reptiles. Rivers were positively correlated only with vulnerability of mammals.

Selection of connectivity areas for the design of a vertebrate conservation system

PACs, the Natura 2000 Network and connectivity areas delineated the spatial extent of the proposed vertebrate conservation planning (Fig. 4.4). It included special elements for conservation, habitats for focal species, and landscape types relevant for biodiversity conservation. All together, they represented ~29% of the Castilla-La Mancha territory.

Based on the location of the least-cost paths, we delineated connectivity areas of under-represented agroecosystems. The spatial extent of the extra patches required to reach the target 15% of habitat representation is shown in Fig. 4.4. After combining identified PACs and the Natura 2000 Network with selected connectivity areas, the new extent of rain-fed cropland represented 15.3% (16296 ha added), mosaic of farms 15.4% (34688 ha added), vineyard 15.8% (15258 ha added), and olive grove 15.9% (8460 ha added). Mosaic of natural vegetation and farm with *dehesa* were land-use types that were under-represented in the Natura 2000 Network, however, it was not necessary to select additional patches for these because the existing patches in combination with PACs extend over an area of ~21% and ~77%, respectively (Table 4.3).

Discussion

Criteria for identifying priority areas for conservation planning

An index to measure diversity, such as the Combined Index of species richness, geographic rarity and level of threat of species present in a given area, has theoretically a notable intrinsic value. Our results confirm the value of the Combined Index. We showed that it was the most effective measure of diversity, by retaining all species and all threatened species of vertebrates within the lowest number of areas. These results fit with our previous studies that used cells of 50 × 50 km (Rey Benayas & de la Montaña 2003) and cells of 20 × 20 km (Rey Benayas *et al*. 2005). Consistency across different scales of analysis significantly increases the robustness of this criterion. Thus, the Combined Index is a useful tool for determining special conservation elements. Undoubtedly, identification of PACs is dependent on the quality of species distribution data (especially for rare species), including locational precision and sampling bias.

Species richness is assumed to be an indicator of conservation value and is typically considered to optimise conservation targets (Prendergast *et al*. 1999; Meir *et al*. 2004). Our current and previous results have shown that both the Combined Index and the rarity criterion are more effective than the richness criterion. This fact has been reported in other works (Margules, Nicholls & Pressey 1988; Haeupler & Vogel 1999). Thus, selecting sites that contain the highest number of species is not the most efficient way to maximally represent biodiversity (Pimm & Lawton 1998; Reid 1998).

Existing conservation areas and priority areas for conservation

It is useful to identify areas with outstanding features of biodiversity to rank priorities for optimising resource investment in conservation. In our study, the Natura 2000 Network considerably improved the guarantees for conservation for all taxonomic groups, since gaps related to PACs decreased significantly with respect to Natural Protected Areas. This was predictable because there was a six-fold increase in the amount of protected area. However, our gap analysis showed that the Natura 2000 Network is insufficient to guarantee the protection of all species in Castilla-La Mancha. One hundred and twenty-one PACs defined by the Standardized Biodiversity Index would be necessary to achieve the desirable protection level, but 14 of these PACs were not included within the Natura 2000 Network.

Gaps between PACs defined by Combined Index for amphibians and the existing protected areas are more numerous than for other *taxa*, as we have found at a lower analysis scale (see Rey Benayas & de la Montaña 2003). Ecological requirements of amphibians contribute to this fact, because they need adequate environmental moisture and specific habitats for reproduction that are scarce in Mediterranean climate regions (Semlitsch 2000; Kiesecker *et al*. 2001; Green 2003). Thus, amphibian populations are frequently concentrated in small and isolated wetlands without protection. Correlations between richness of amphibians and amount of *dehesa*, grassland and farm with *dehesa*, which are habitats with small seasonal wetlands of natural origin or man-made for cattle use, support this hypothesis.

The Natura 2000 Network in Castilla-La Mancha region satisfactorily represents forests, shrublands, grasslands and wetlands at a landscape scale. However, *dehesa* is the only agroecosystem represented adequately. Traditional farm of rain-fed cropland, olive grove and vineyard, and areas of mosaic of farms, mosaic of natural vegetation and farm with *dehesa* are all under-represented, as is their biodiversity. This landscape types form agroecosystems with high landscape heterogeneity and habitat diversity that are very important for wildlife conservation (Farina 1997; Tucker 1997; Benton *et al*. 2003).

Agroecosystems are often considered habitats with low conservation value. However, the historical loss of many natural ecosystems has caused the inhabitant species to become strongly dependent on their secondary agricultural habitats (Kleijn *et al*. 2006). Subsequent agricultural changes in Europe have caused loss of biodiversity in most agroecosystems (Benton *et al*. 2002). This decline in recent decades has been well documented for farmland birds (Tucker & Heath 1994; Chamberlain *et al*. 2000; Birdlife 2004; Heer *et al*. 2005; Wretenberg *et al*. 2006). Ours results are consistent with the importance of these agroecosystems for threatened birds.

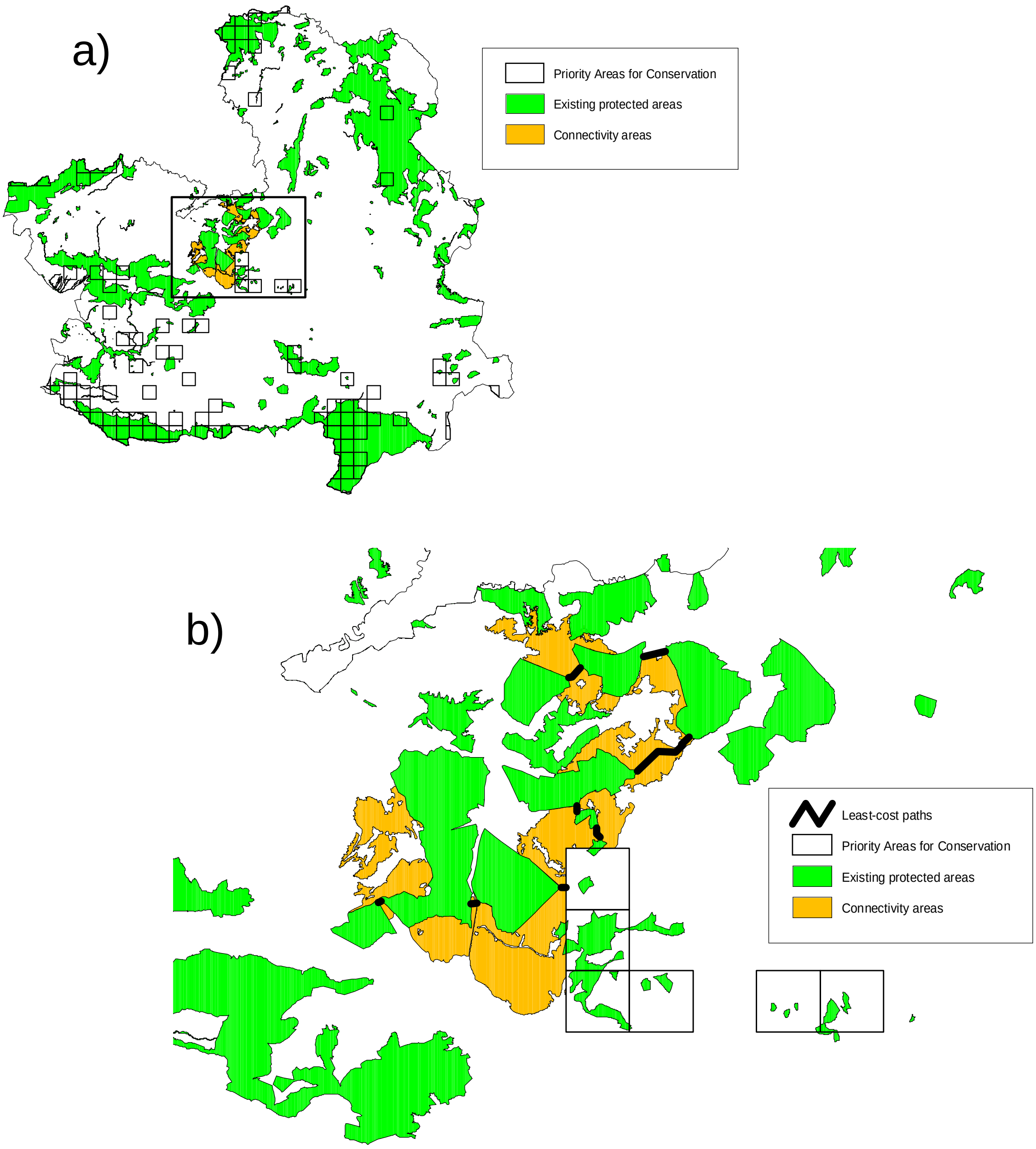


Figure 4.4. (a) Spatial distribution of important vertebrate diversity areas for systematic conservation planning in Castilla-La Mancha, including Priority Areas for Conservation, existing protected areas and connectivity areas. (b) Higher magnification of the boxed area in (a). This shows the largest and nearest target areas selected after applying the nearest features extension of ArcView 3.2, which allowed selection of additional patches of under-represented land-use types for connectivity.

Vulnerability of birds is the criterion to identify PACs that presents the highest positive correlation with the extent of the three agroecosystems that are dominant in the study area (rain-fed cropland, mosaic of farms and vineyard).

Traditional landscapes of farmland and extensively managed mosaics are characteristic of Mediterranean regions. They are inhabited by many rare and threatened species (Suárez *et al*. 1997). For example, after losing most of the primary habitat of natural steppes, the major conservation problem for steppe birds is the loss and degradation of traditional farmland habitats and pseudo-steppes, due to intensification (irrigated lands, glasshouses), abandonment (shrubland growth) and sectorial politicies (infrastructures) (Tucker & Heath 1994; Siriwardena *et al*. 1998; Donald *et al*. 2001; Brotons *et al*. 2004; Verhulst *et al*. 2004). Most of European steppe birds populations are distributed in Spain. We list these species (included in Annex I of Birds Directive) considering the European endangered status (SPEC scores), obtained from Tucker & Heath (1994). The percentage of European population distributed in Spain are: 100% of Dupont's Lark (*Chersophilus duponti*) (SPEC-3), >90% of Short-toed Lark (*Calandrella brachydactyla*) (SPEC-3), Calandra Lark (*Melanocorypha calandra*) (SPEC-3), Black Wheatear (*Oenanthe leucura*) (SPEC-3), Great Bustard (*Otis tarda*) (SPEC-1), Pin-tailed Sandgrouse (*Pterocles alchata*) (SPEC-3), Black-beilled Sandgrouse (*Pterocles orientalis*) (SPEC-3) and Little Bustard (*Tetrax tetrax*) (SPEC-1), >75% of Stone-curlew (*Burhinus oedicnemus*) (SPEC-3), Black-shouldered Kite (*Elanus caeruleus*) (SPEC-3), Lesser Krestel (*Falco naumanni*) (SPEC-1) Thekla Lark (*Galerida theklae*) (SPEC-3), and others. Furthermore, they are areas with large numbers of extensive sheep and cattle, which provide the necessary carrion to rare raptors species like Monk Vulture (*Aegypius monachus*) (SPEC-1) and Egyptian Vulture (*Neophron percnopterus*) (SPEC-3).

Proposal for conservation planning

Our assessment shows that approximately 29% of Castilla-La Mancha land is required to protect special conservation elements, focal species and all landscape types. This agrees with other studies that estimate that the proportion of a region required to capture important elements of biodiversity is between 33 and 75% (see Soulé & Sanjayan 1998 for review).

Our proposal achieved two conservation targets: protection for 100% of species and representation of all landscapes types, which is a common limitation in multi-track approaches applied across large regions. Thus, the combination of PACs, the existing Natura 2000 Network and new connectivity areas, results in a spatial configuration that achieves the first objective of nature reserves: to represent the biodiversity of each region (Margules & Pressey 2000). However representation of biodiversity does not guarantee the persistence of viable population (the second objective of reserves) or the protection of ecological processes that maintain biodiversity (Salomon *et al*. 2006). Targets for off-reserve conservation are particularly important, and conservation on private land is also essential (Soares-Filho *et al*. 2006), especially in fragmented and humanised landscapes, in which reserves are likely to be small and isolated.

Currently, many species depend on large areas of traditional agriculture. Therefore, our proposed conservation planning includes additional areas of under-represented agroecosystems that improve connectivity into protected area networks for strengthening biodiversity conservation. Furthermore, to protect farmland wildlife adequately, it is necessary to improve agri-environment schemes (Kleijn & Sutherland 2003; Kleijn *et al*. 2006), which are considered the most important policy instrument for protecting biodiversity in agricultural landscapes (EEA 2004). This should avoid unsustainable intensive farming that is damaging for biodiversity conservation and rural economies.

Presence/absence data are frequently used in approach at regional scale (Lennon *et al*. 2001; Manley *et al*. 2004; Bonn & Gaston 2005) and the value of diversity measures based on such data has been questioned for some authors in landscape planning (Smith & Wilson 1996; Stirling & Wilsey 2001). Our approach provides useful information, but we must consider that our results are scale dependent. We found negative correlations between the Standardized Biodiversity Index and the extent of agroecosystems, despite a large increase in the protected area of these habitats when we incorporated PACs into conservation planning design. This could be explained in part by bias that arises from agroecosystems being the more extensive habitats in the study area but with lower percentage representation. It may also be because local-scale considerations are not adequately captured in broad-scale assessments (Rouget 2003). In Castilla-La Mancha, wetlands and lagoons are frequently small and disperse, but with high biodiversity, especially when they are surrounded by drylands. This fact contributes to increase agroecosystems bias in the protected area network, since some PACs have been probably established because they include some wetland or lagoon among a big proportion of agroecosystems. Positive correlation between the SBI and wetlands supports this hypothesis. This was predictable because SBI is designed to establish relevant areas of biodiversity and not to maximise representation of landscape type.

Future research should apply specific species analysis (rare or threatened species), incorporating habitat suitability and population viability for optimal selection of core areas (e.g. Beazley *et al*. 2005). We suggest a similar approach to establish adequate ecological restoration and environmental impact mitigation, and to integrate social and economic considerations. Land protection is often driven by local opportunities and politics rather than by *a priori* assessment of ecological value. But, in order to progress towards the global target of reducing the current rate of biodiversity loss by 2010 (UNEP 2002), we need strategies for managing whole landscapes including areas allocated to both production and conservation. In humanised landscapes, it is of fundamental importance to maintain traditional resources management (extensive cattle, rotation of farmland, and exploitation of native timber species) that is the origin and future of biodiversity in these areas.

In conclusion, we found that: (1) the Combined Index is an effective and robust measure of diversity; (2) the Natura 2000 Network delivers benefits for biodiversity conservation in Castilla-La Mancha, but represents insufficiently the most traditional agricultural habitats and does not guarantee the protection of their threatened vertebrate species, especially birds; and (3) our three-track approach achieves representation of every landscape and vertebrate diversity in the region, and despite its limitations, has the potential for application in other regions.

Acknowledgments

This study was funded by the grant projects REN2000-745 and CGL2004-355-BOS (CICYT, Ministry of Education and Science, Spain). E.M. received research fellowships from the University of Alcalá and the Madrid Autonomous Community. We are grateful to I. Valencia for collaborating in this research. R. Sandoval and F. Alburquerque kindly improved the English of the manuscript.

References

Andelman, S. J., and M. R. Willig. 2003. Present patterns and future prospects for biodiversity in the Western Hemisphere. Ecology Letters 6: 818-824.

Beazley, K., L. Smandych, T. Snaith, F. Mackinnon, Jr. P. Austen-Smith, and P. Duinker. 2005. Biodiversity considerations in conservation system planning: map-based approach for Nova Scotia, Canada. Ecological Applications 15: 2192-2208.

Benton, T. G., J. A. Vickery, and J. D. Wilson. 2003. Farmland biodiversity: is habitat heterogeneity the key? Trends in Ecology and Evolution 18: 182-188.

BirdLife International. 2004. Birds in Europe: Population Estimates, Trends and Conservation Status. BirdLife Conservation Series No. 12, BirdLife International, Cambridge.

Bonn, A., and K. J. Gaston. 2005. Capturing biodiversity: selecting priority areas for conservation using different criteria. Biodiversity and Conservation 14: 1083-1100.

Brotons, Ll., S. Mañosa, and J. Estrada. 2004. Modelling the effects of irrigation schemes on the distribution of steppe birds in Mediterranean farmland. Biodiversity and Conservation 13: 1039-1058.

Cabeza, M., and A. Moilanen. 2001. Design of reserve network and the persistence of biodiversity. Trends in Ecology and Evolution 16: 242-248.

Chamberlain D.E., R. J. Fuller, R. G. H. Bunce, J. C. Duckworth, and M. Shrubb. 2000. Changes in the abundance of farmland birds in relation to the timing of agricultural intensification in England and Wales. Journal of Applied Ecology 37: 771-788.

Cowling, R. M., and R. L. Pressey. 2003. Introduction to systematic conservation planning in the Cape Floristic Region. Biological Conservation 112: 1-13.

Cowling, R. M., R. L. Pressey, M. Rouget, and T. Lombart. 2003. A conservation plan for a global biodiversity hotspot: the Cape Floristic Region, South Africa. Biological Conservation 112: 191-216.

Donald, P. F., R. E. Green, and M. F. Heath. 2001. Agricultural intensification and the collapse of the Europe's farmland bird population. Proceedings of the Royal Society of London Ser. B. Biological Sciences 268: 25-29.

EEA. 2004. High Nature Value Farmland - Characteristics, Trends and Policy Challenges. European Environment Agency, Copenhagen.

ESRI (Environmental Systems Research Institute). 1999. ArcView. Version 3.2. ESRI, Redlands, California.

European Commission. 1993. EUR 12585 - CORINE land cover map and technical guide. Office for Official Publications of the European Communities, Luxembourg.

Farina, A. 1997. Landscape structure and breeding bird distribution in a sub-Mediterranean agro-eco system. Landscape Ecology 12: 365-378.

Gaston, K. J., et al. 2006. The ecological effectiveness of protected areas: The United Kingdom. Biological Conservation 132, 76-87.

Green, D. M. 2003. The ecology of extinction: population fluctuation and decline in amphibians. Biological Conservation 111: 331-343.

Groppalli, R. 1993. Breeding birds in traditional tree rows and Hedges in the central Po Valley (Province of Cremona, Northern Italy). Pages 153-158 in R. G. H. Bunce, L. Ryszkowski and M. G. Paoletti, editors. Landscape Ecology and Agroecosystems. Lewis Publishers, Boca Raton, FL.

Haeupler, H., and A. Vogel. 1999. Plant diversity in Germany: A second review. Acta Botanica Fennica 162: 55-59.

Heer, M. de, V. Kapos, and B. J. E. ten Brink. 2005. Biodiversity trends in Europe: development and testing of a species trend indicator for evaluating progress towards the 2010 target. Philosophical Transactions of the Royal Society Ser. B. Biological Science 360: 297-308.

IUCN. 2001. IUCN Red List Categories and Criteria: Version 3.1. IUCN Species Survival Commission. IUCN, Gland, Switzerland and Cambridge. ii + 30 pp.

IUCN. 1994. Guidelines for Protected Area Management Categories. IUCN, Gland, Switzerland and Cambridge.

Jenness, J. 2007. Nearest features (nearfeat.avx) extension for ArcView 3.2, v. 3.8b. Jenness Enterprises. Available from http://www.jennessent.com

Kiesecker, J. M., A. R. Blaustein, and L. K. Belden. 2001. Complex causes of amphibians population declines. Nature 410: 681-684.

Kleijn, D., et al. 2006. Mixed biodiversity benefits of agri-environment schemes in five European countries. Ecology Letters 9: 243-254.

Kleijn, D. and W. J. Sutherland. 2003. How effective are European agri-environment schemes in conserving and promoting biodiversity? Journal of Applied Ecology 40: 947-969.

Lennon, J. J., P. Koleff, J. J. D. Greewood, and K. J. Gaston, K.J. 2001. The geographical structure of British bird distributions: diversity, spatial turnover and scale. Journal of Animal Ecology 70: 966-979.

Manley, P.N., W. J. Zielinski, M. D. Schlesinger, and S. R. Mori. 2004. Evaluation of a multiple-species approach to monitoring species at the ecoregional scale. Ecological Applications 14: 296-310.

Margules, C. R., and R. L. Pressey. 2000. Systematic conservation planning. Nature 405: 243-253.

Margules, C. R., A. O. Nicholls, R. L. and Pressey. 1988. Selecting networks of reserves to maximize biological diversity. Biological Conservation 43: 63-76.

Meir, E., S. Andelman, and H. P. Possingham. 2004. Does conservation planning matter in a dynamic an uncertain world? Ecology Letters 7: 615-622.

Ministerio de Medio Ambiente. 2002a. Atlas y libro rojo de los anfibios y reptiles de España, Madrid.

Ministerio de Medio Ambiente. 2002b. Atlas de los mamíferos terrestres de España, Madrid.

Ministerio de Medio Ambiente. 2003. Atlas de las aves reproductoras de España, Madrid.

Myers, N., R. A. Mittermeler, C. G. Mittermeler, G. A. B. da Fonseca, and J. Kent. 2000. Biodiversity hotspots for conservation priorities. Nature 403: 853-858.

Oliver, I., et al. 2004. Land systems as surrogates for biodiversity in conservation planning. Ecological Applications 14: 485-503.

Pimm, S. L. J. H. and Lawton. 1998. Planning for biodiversity. Science 279: 2068-2069.

Pimm, S.L., et al. 2001. Can we defy nature's end? Science 293: 2207-2208.

Prendergast, J.R., R. M. Quinn, and J, H. Lawton. 1999. The gaps between theory and practice in selecting nature reserves. Conservation Biology 13: 484-492.

Ray, N. 2005 PathMatrix: a GIS tool to compute effective distances among samples, Molecular Ecology Notes 5: 177-180.

Reid, W. V. 1998. Biodiversity hotspots. Trends in Ecology and Evolution 13: 275-280.

Rey Benayas, J. M., and E. de la Montaña. 2003. Identifying areas of high-value vertebrate diversity for strengthening conservation. Biological Conservation 114: 357-370.

Rey Benayas, J. M., E. de la Montaña, J. Belliure, and X. R. Eekchout. 2006. Identifying areas of high herpetofauna diversity that are threatened by planned infrastructure projects in Spain. Journal of Environmental Management 79: 279-289.

Rodrigues, A.S., et al. 2004. Effectiveness of the global protected area network in representing species diversity. Nature 428: 640-643.

Rouget, M. 2003. Measuring conservation value at fine and broad scales: implications for a diverse and fragmented region, the Agulhas Plain. Biological Conservation 112: 217-232.

Salomon, A. K., J. L. Ruesink, and R. E. DeWreede. 2006. Population viability, ecological processes and biodiversity: Valuing sites for reserve selection. Biological Conservation 128: 79-92.

Scott, J. M., F. W. Davis, R. G. McGhie, R. G. Wright, C. Groves, and J. Estes. 2001. Nature reserves: Do they capture the full range of America's biological diversity? Ecological Applications 11: 999-1007.

Semlitsch, R. D. 2000. Principles for management of aquatic-breeding amphibians. Journal of Wildlife Management 64: 615-631.

Seoane, J., J. H. Justribó, F. García, J. Retamar, C. Rabadán, and J. C. Atienza. 2006 Habitat-suitability modelling to assess the effects of land-use changes on Dupont's lark *Chersophilus duponti*: A case study in the Layna Important Bird Area. Biological Conservation 128: 241-252.

Siriwardena, G. M., S. R. Baillie, S. T. Buckland, R. M. Fewster, J. M. Marchant, and J. D. Wilson. 1998. Trends in the abundance of farmland birds: a quantitative comparision of smoothed Common Bird Census indices. Journal of Applied Ecology 35: 24-43.

Smith, B., and J. B. Wilson. 1996. A consumer's guide to evenness indices. Oikos 76: 70-82.

Soares-Filho, B. S. 2006. Modelling conservation in the Amazon basin. Nature 440: 520-523.

Soulé, M. E. 1991. Conservation: Tactics for a constant crisis. Science 253: 744-750.

Soulé, M. E., and M. A. Sanjayan. 1998. Conservation targets: Do they help? Science 279: 2060-2061.

Stirling, G. and B. Wilsey. 2001. Empirical relationships between species richness, evenness, and proportional diversity. American Naturalist 158: 286-299.

Suárez, F., M. A. Naveso, and E. De Juana. 1997. Farming in the dryland in Spain: birds of the pseudosteppes. Pages 297-330 in D. J. Pain and M. W. Pienkowski, editors. Farming and Birds in Europe: The Common Agricultural Policy and its Implication for Birds Conservation. Academic Press, London.

Tucker, G. M., and M. F. Heath. 1994. Birds in Europe: Their Conservation Status. BirdLife International, Cambridge, UK.

Tucker, G. M. 1997. Priorities for birds conservation in Europe: the importance of the farmed landscape. Pages 79-116 in D. J. Pain and M. W. Pienkowski, editors. Farming and Birds in Europe: The Common Agricultural Policy and its Implication for Birds Conservation. Academic Press, London.

Underwood, E. C., M. R. Shaw, K. A. Wilson, P. Kareiva, K. R. Klausmeyer, M. F. McBride, M. Bode, S. A. Morrison, J. M. Hoekstra, and H. P. Possingham. 2008. Protecting biodiversity when money matters: maximizing return on investment. PLoS ONE 3: e1515. doi:10.1371/journal.pone.0001515

UNEP (2002) "Report on the Sixth Meeting of the Conference of the Parties to the Convention on Biological Diversity (UNEP/CBD/COP/6/20/Part 2) Strategic Plan Decision VI/26"; available from www.biodiv.org/doc/meetings/cop/cop-06/official/cop-06-20-part2-en.pdf.

Verhulst, J., A. Báldi, and D. Kleijn. 2004. Relationship between land-use intensity and species richness and abundance of birds in Hungary. Agriculture, Ecosystems and Environments 104: 465-473.

Willianson, T. M. 1986. Parish boundaries and early fields: continuity and discontinuity. Journal of History Geographic 12: 241-248.

Wretenberg, J., A. Lindstrom, S. Svensson, T. Thierfelder, and T. Part. 2006. Population trends of farmland birds in Sweden and England: similar trends but different patterns of agricultural intensification. Journal of Applied Ecology 43: 1110-1120.

Capítulo 5

Los árboles que no dan flores dan nidos;
y un nido es una flor con pétalos de pluma;
un nido es una flor color de pájaro;
cuyo perfume entra por los oidos.
Los árboles que no dan flores dan nidos.

Fernán Silva Valdés

Folio con 24 aves de la Paráfrasis de los Ornithiakas de Dionysios. Siglo I.

Capítulo 5

Respuesta de la comunidad de aves al resalveo del maquis mediterráneo

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

Enrique de la Montaña, José M. Rey Benayas, Luis M. Carrascal (2006). Response of bird communities to silvicultural thinning of Mediterranean maquis. Journal of Applied Ecology 43, 651-659.

Resumen

1. La gestión forestal debería considerar la conservación de la biodiversidad. Los propietarios de tierras en algunas regiones mediterráneas europeas reciben subsidios por el resalveo de las zonas de maquis denso. Esta práctica elimina la mayoría de los matorrales y los árboles de menores dimensiones, mientras que los árboles más altos son podados para producir masas forestales más abiertas. Se ha investigado el impacto de esta práctica en el estado de conservación de la comunidad de aves.
2. Se diseñó un "experimento natural" a gran escala en España central, que incluía 21 lugares con maquis sometidos a resalveo y otros tantos que no. Cada lugar fue muestreado mediante cinco puntos de conteo, consistentes en una parcela de 50 m de radio, en dos años consecutivos y en invierno y primavera. La estructura de la vegetación fue muestreada después de los censos de aves en parcelas de 10 m de radio que coincidían con los centros de los puntos de conteo de aves. Los análisis de datos se realizaron mediante ANOVAs de medidas repetidas.
3. El resalveo produjo un incremento significativo de la riqueza de aves, pero no tuvo ningún efecto en la densidad total de aves. La masa corporal media de las especies de los lugares resalveados fue significativamente mayor que en los lugares de maquis, con mayor densidad de vegetación, no resalveados. La densidad de las especies de aves que se alimentan en el suelo fue indistinguible entre los lugares con y sin resalveo, mientras que la densidad de las especies que se alimentan en el follaje fue mayor en las zonas sin resalveo. La densidad invernal de especies granívoras fue marginalmente superior en zonas resalveadas, mientras que las especies frugívoras e insectívoras fueron marginalmente más abundantes en las zonas no resalveadas.
4. Las zonas resalveadas presentaron mayores densidades de especies cuyo estado de conservación en Europa es de alta preocupación. Las densidades invernales de especies cinegéticas fueron también mayores en estas zonas.
5. *Síntesis y aplicaciones*: Esta es la primera vez que una manipulación experimental a gran escala de la estructura del hábitat y del volumen de la vegetación, ha demostrado el predicho efecto alométrico de la complejidad estructural del hábitat en la masa corporal media de una comunidad de aves. Los resalveos de zonas forestales mediterráneas densas aumentaron la heterogeneidad del hábitat y su idoneidad para varias especies de aves e incrementaron la riqueza de especies. También fueron beneficiosos para especies de interés en conservación y para especies de aves cinegéticas no amenazadas. Sin embargo, deberían preservarse algunas áreas sin resalveo como refugio para las pocas especies que se ven afectadas por los resalveos.

Palabras clave: Aves cinegéticas; Densidad; Estado de conservación; Estructura de la vegetación; Gremios; Masa corporal; Riqueza de especies

Response of bird communities to silvicultural thinning of Mediterranean maquis

Enrique de la Montaña[1], José M. Rey-Benayas[1], Luis M. Carrascal.[2]

[1] *Departamento de Ecología, Universidad de Alcalá. Alcalá de Henares, 28871 Madrid, Spain.*

[2] *Dpto. Biodiversidad y Biología Evolutiva, Museo Nacional de Ciencias Naturales, CSIC, 28006 Madrid, Spain.*

Abstract

1. Woodland management should consider biodiversity conservation worldwide. Land owners in some European Mediterranean regions receive subsidies to thin dense maquis. This practice consists of the elimination of most shrubs and saplings, and the pruning of the tallest trees to favour more opened woodland stands. We investigated how this practice affects the structure of bird communities to assess their conservation.
2. We designed a large scale 'natural experiment' that included 21 paired thinned and un-thinned maquis stands in Central Spain. Every stand was sampled by means of 5 point counts, each consisting of a 50 m radius plot, in two consecutive years and in winter and spring. The vegetation structure was characterized after bird censuses in 10 m radius plots that coincided with the centres of the bird point counts. Data analyses were based on repeated-measures ANOVAs.
3. Thinning was responsible for a significant increase in species richness, but did not have any effect on total bird density. Average body mass of species in thinned stands was significantly larger than in un-thinned, more densely vegetated, stands. Density of ground searchers was undistinguishable in thinned and in un-thinned stands, whereas density of foliage gleaners was higher in un-thinned stands. Winter density of granivorous species was marginally higher in thinned stands, whereas insectivorous and frugivorous species were marginally more abundant in un-thinned stands.
4. Thinned areas were occupied by higher densities of bird species whose European conservation status is of high concern. Winter density of game birds was also higher in thinned stands.
5. *Synthesis and applications.* This is the first time that a large-scale experimental manipulation of habitat structure and vegetation volume has demonstrated the predicted allometric effect of habitat structural complexity on the average body mass of a bird community. Thinning of dense Mediterranean woodland enhances habitat heterogeneity and suitability for several bird species and increased species richness. This practice was also beneficial for species of conservation concern and non-threatened gamebirds. However, some un-thinned patches should be preserved to provide refuge for the few species that are impacted by thinning.

Keywords: Body mass; Conservation status; Density; Gamebirds; Guilds, Species richness; Vegetation structure

Introduction

It has long been recognized that structural complexity of vegetation influences the structure of bird communities, including the number and diversity of niches, local abundance, and guilds or groups of species that exploit similar resources (Wiens 1989; Díaz *et al.* 1998). Human activities may profoundly modify land cover and the architecture of vegetation and, consequently, may affect the composition and abundance of bird species (Blondel & Aronson 1999; Heikkinen *et al.* 2004). Particularly, in many Mediterranean regions there has been a secular clearing and thinning of the Mediterranean maquis that has turned vast expanses of dense shrublands and woodlands into more open forests or man-made savannas such as the 'dehesas' (Grove & Rackham 2001; Maldonado *et al.* 2002). In this study, we investigate for the first time how the thinning of the Mediterranean maquis affects bird communities.

Since 1990, land owners in some European Mediterranean regions have received subsidies to thin dense maquis. The subsidies aim to provide environmental services and local income. This practice consists of the eliminating most shrubs and saplings, and pruning the tallest trees to favour more opened woodland stands. Thinning reduces competition among overstorey trees and increases penetration of solar radiation to the forest floor, thereby increasing the growth, size, branch diameter, and crown ratio of remaining trees and stimulating the development of understorey vegetation (Ducrey & Toth 1992; Perevolotsky & Haimov 1992; Kaplan & Gutman 1996; Homyack, Harrison & Krohn 2004). This type of management aims to prevent fire, improve habitat conditions for game mammals (e.g. rabbit, deer) and, to a lesser extent, create pastures for extensive sheep and cattle grazing, and produce firewood. It may also increase structural complexity and improve the habitat for several vertebrate species of wildlife (Sullivan, Sullivan & Lindgren 2001; Patriquin & Barclay 2003). The potential of thinning to redirect the developmental trajectory of young stands towards stands with a higher structural diversity has gained overall recognition during the last decade (DeBell *et al.* 1997).

Several studies in different ecosystems have addressed the question of how forest management affects vegetation structure (Marañón *et al.* 1999; Thomas, Halpen & Falk 1999) and the associated effects on the community structure of different taxonomic groups (Sullivan, Lautenschlager & Wagner 1999; Halaj, Ross & Moldenke 2000; Liow, Sodhi & Elmqvist 2001; Sullivan *et al.* 2001; Thompson, Baker & Ter-Mikaelian 2003), chiefly bird communities (Hagar, Howlin & Ganio 2004; Sekercioglu 2002; Hayes, Weikel & Huso 2003). These studies have found that the responses of bird communities to vegetation thinning are generally complex (Carey 2003), and positive effects on species richness, abundance of some species, and an absence of species extirpation from thinned stands, are usually described. However, the effects of thinning of the Mediterranean maquis on bird communities have not been previously addressed, although Sánchez-Zapata & Calvo (1999) have pointed out that habitat mosaics created by forestry and traditional farming are especially important for Mediterranean raptors.

Our major objective was to ascertain the effects of thinning the Mediterranean maquis on species richness, composition, abundance, and conservation status of bird communities.

We considered functional guilds of species according to their habitat use and trophic preferences by designing a large scale 'natural experiment' that included paired thinned and un-thinned maquis stands in Central Spain. We also examined whether the effects of thinning demonstrated previously in other forest environments can be extended to the Mediterranean region. Further, we ask whether these effects depend upon the biogeographic origin and habitat requirements of its avifauna (Suárez-Seoane, Osborne & Baudry 2002). Finally, the experimental manipulation of vegetation by thinning allowed us to test the prediction of the macroecological hypothesis previously examined in a phylogenetic context (Polo & Carrascal 1999; Gaston & Blackburn 2000): the structural complexity and vegetation density of the habitat should act as a selective filter of the species body masses, favouring settlement by small-sized species in complex and dense habitats due to the manoeuvrability constraints.

Materials and methods

Study area

Field work was conducted in 21 maquis stands in Ciudad Real province located in the southern Spanish plateau. The study area is 19749 km^2 in extent and mid-coordinates are 38.9º north and 3.8º west. Altitude ranges between 423 and 999 m. Climate is continental Mediterranean, with dry, hot summers and cold winters. Mean annual temperature and total annual precipitation are 13.7 ºC and 510 mm, respectively. An increasing aridity gradient can be recognized from northwest to southeast, annual rainfall ranging between 700 and 300 mm. The potential vegetation of most stands is dominated by the evergreen holm oak *Quercus ilex* L. However, land use has transformed these vast evergreen woodlands into a mosaic of patches dominated by woodland remnants, shrubland (e.g. *Q. coccifera* L., *Erica arborea* L., *Cistus ladanifer* L. and *Rosmarinus oficinalis* L.), pasture lands and cropland. There is a shortage of fruit trees and shrubs in the area.

Bird census

We selected 21 localities distributed throughout the study area to reflect its environmental heterogeneity, chiefly the aridity gradient. Each locality included a pair of maquis stands (thinned and un-thinned). A thinned stand had to be at least 12 ha to be sampled; they averaged 19.9 ± 12.4 ha in extent. In the selected thinned stands, thinning was practiced between 2 and 10 years before this study (4.1 ± 2.1 years). Stands were thinned by their land owners to favour cattle grazing and hunting, to reduce the risk of fires, and to sell firewood. As close as possible to the selected thinned stands, and always within a 3 km radius, we selected an un-thinned maquis stand with similar physical features including orientation, slope and soil type. Thus, the un-thinned stands act as reference stands that resemble the features of the thinned stands prior to their intervention.

Every stand was sampled by means of five point counts separated 100 m from each other with each count lasting ten minutes. The first and fifth point count was placed at least 100 m away from the stand edge. In each point count we established a circular plot of 50 m radius (Bibby *et al.* 2000) where we noted the presence and abundance of every bird species (Appendix 5.A). Point count is the best census method in this habitat because un-thinned stands are very dense and difficult to traverse.

Nearly all detections were auditive (singing males, alarm calls, etc.), so bird detection was not dependent on vegetation obstruction or visibility. This method does not provide absolutes densities, but relative abundances. Nevertheless, the small area covered by the plots (0.78 ha), and the relatively long time devoted to bird counts, maximizes the detection probability of species and, thus, the accurate estimations of densities (Shiu & Lee 2003). Additionally, preliminary census of the bird community and to evaluate sampling efficiency, indicated that our conclusions are not biased by detectability problems. Species richness was assessed by considering all species contacted in the pooled sample of the five 10 minutes stations, in order to include scarce species that would not normally be detected.

Censuses started at sunrise and lasted for ca. four hours. Sampling was carried out during the winter (December and January) and the breeding season (May and the first fortnight of June) in two consecutive years (2002 and 2003). All samples were obtained on rainless and nearly windless days (wind speed < 3 m/s) to reduce detectability problems. All censuses were conducted by the same researcher (EM).

Vegetation structure

The vegetation structure was characterized after bird censuses in 10 m radius plots that coincided with the centres of the bird point counts. Habitat structure for each stand was measured using the average of the five 10 m plots. All habitat measurements (Table 5.1) were carried out by the same observer in order to avoid interpersonal bias.

Species' characteristics

Data on body size, winter diet and main foraging substrata were obtained from Perrins (1998). The body mass of each species was calculated by averaging the data for adults of

Table 5.1. Measured variables and summary statistics that describe the structure of the vegetation in n = 21 localities, comparing thinned (THIN) vs. un-thinned or reference (REF) maquis stands. SE: standard error. Sample size is n = 21. Degrees of freedom for repeated measure ANOVAs are 1, 20.

	Mean		SE			
	THIN	REF	THIN	REF	*F*	*P*
Cover of bare ground (%)	22.1	26.3	4.4	4.9	0.79	0.385
Cover of moss (%)	9.9	12.5	2.6	3.1	4.04	0.058
Cover of herbaceous plants (%)	42.7	24.0	7.2	6.3	11.20	0.003
Cover of shrubs less than 25 cm tall (%)	10.0	6.2	2.0	1.5	3.36	0.082
Cover of shrubs 25-50 cm tall (%)	10.4	5.3	2.0	1.0	7.00	0.015
Cover of shrubs 50-100 cm tall (%)	7.4	7.2	2.0	1.6	0.01	0.947
Cover of shrubs 100-150 cm tall (%)	4.9	7.3	2.3	2.0	4.70	0.042
Cover of shrubs more than 150 cm tall (%)	6.8	20.7	2.8	5.1	6.76	0.017
Average height of shrubs (cm)	58.3	93.2	7.0	9.6	10.20	0.005
Tree crown cover (%)	24.6	36.9	2.7	4.2	9.34	0.006
Average height of trees (m)	3.7	3.8	0.2	0.2	0.00	0.998
Number of trunks less than 5 cm d.b.h. (no. ha^{-1})	181.1	651.6	31.6	79.2	32.40	0.000
Number of trunks 5-10 cm d.b.h. (no. ha^{-1})	283.3	351.7	50.8	77.7	0.85	0.367
Number of trunks 10-20 cm d.b.h. (no. ha^{-1})	67.5	56.0	13.3	14.4	0.78	0.387
Number of trunks 20-30 cm d.b.h. (no. ha^{-1})	6.7	6.0	2.9	1.9	0.05	0.823
Number of trunks more than 30 cm d.b.h. (no. ha^{-1})	2.5	3.8	1.2	1.5	0.70	0.411

both sexes of the subspecies that inhabit the study region. The average body mass of bird species in each census plots was calculated by means of weighted averages, using the species densities in each locality-stand as weights. The assignation of each species to an ecological category was made according to gross descriptions pertaining to food habits and spatial niche. Birds were grouped into seed-eaters, insectivorous and frugivorous species when, respectively, seeds, arthropods and fruits were the main constituents of the winter diet. The diet during the breeding season was not considered because all the species inhabiting the study area are mainly consume arthropods during this time of the year. The spatial niche of species was categorized in the following groups: ground searchers (those mainly foraging on the ground, in the herbaceous layer or among leaf litter) and foliage gleaners (those mainly foraging among the leaves, twigs or small branches of trees or shrubs). Other spatial niche groups were omitted due to their under-representation in the sample (e.g. trunk searchers, aerial foragers, avian or mammal predators). For some bird species these categories were inadequate because the birds foraged on several substrata. In those cases, the density of the species was assigned proportionally to each spatial niche group or diet category.

The European endangered status of each species was obtained from Tucker & Heath (1994) using the Species of European Conservation Concern (SPEC) scores. However, this index of conservation concern only refers to breeding populations and many of the sampled species do not have SPEC scores as they are not considered of conservation concern. Twenty-six species were included in the category SPEC-4 because their global populations are concentrated in Europe but they have a favourable conservation status. Fifteen species were assigned to the SPEC category 3, denoting those birds whose global populations are not endangered but have an unfavourable conservation status in Europe. Finally, eight species were considered of conservation concern because their populations were concentrated in Europe and have an unfavourable conservation status (SPEC-2). SPEC categories were scored as follows: Non-SPEC = 5; SPEC-4 = 4; SPEC-3 = 3; SPEC-2 = 2. The most abundant species within this last category in the studied stands were the red-legged partridge *Alectoris rufa* L., green woodpecker *Picus viridis* L., woodlark *Lullula arborea* L., black-eared wheatear *Oenanthe hispanica* L., dartford warbler *Sylvia undata* Boddaert and woodchat shrike *Lanius senator* L. The average values of European conservation status (SPEC score) were calculated by means of weighted averages, using the species densities in each locality-stand as weights.

The influence of maquis thinning on gamebirds was analyzed for these species of hunting interest: red-legged partridge *Alectoris rufa*, quail *Coturnix coturnix* L., woodpigeon *Columba palumbus* L., turtle dove *Streptopelia turtur* L., song thrush *Turdus philomelos* Brehm, and mistle thrush *Turdus viscivorus* L. These species are hunted only in the winter season.

Data analysis

Data analyses were carried out by means of repeated-measures ANOVAs because of the paired design of our natural experiment. Two factors were included in the analyses: (1) thinned vs. un-thinned treatments, and (2) seasonal differences (winter vs. spring censuses).

The interaction term thinning x season was also included in the two-way ANOVAs to test if the effects of thinning were persistent across seasons. One-way repeated measures ANOVAs were only possible in one season for winter trophic groups, winter density of game birds and the SPEC scores for breeding populations. All statistical tests used the means per stand as the sample unit (i.e. the average of census samples per year in two consecutive years).

The time elapsed between the years when thinning was practiced and the years when birds were censused were not homogeneous in the 21 study localities (see above). The influence of this effect on thinning was tested by means of the partial correlation between the measures of vegetation structure in thinned stands and the time elapsed, controlling for the effect of the measures in un-thinned stands. No one of the partial correlations for the 16 habitat variables was significant ($P > 0.107$). Therefore, the heterogeneity across the 21 localities in the time elapsed since thinning does not introduce any bias in the comparisons of thinned vs. un-thinned stands. All statistical analyses were carried out using STATISTICA 6.0 (StatSoft 2001).

Results

Effects of thinning on vegetation structure

Thinning reduced the cover of shrubs taller than 100 cm, average height of shrubs, cover of trees and number of trunks less than 5 cm d.b.h. (Table 5.1). Conversely, it increased the cover of herbaceous vegetation and of shrubs shorter than 50 cm.

Effects of thinning on bird community

Considering both study seasons together, 11 common (> 1 bird 10 ha^{-1}) species were at least 33% more abundant in thinned stands than in un-thinned maquis: corn bunting *Miliaria calandra* L., woodchat shrike *Lanius senator* and thekla lark *Galerida theklae* Brehm and bee-eater *Merops apiaster* L. in

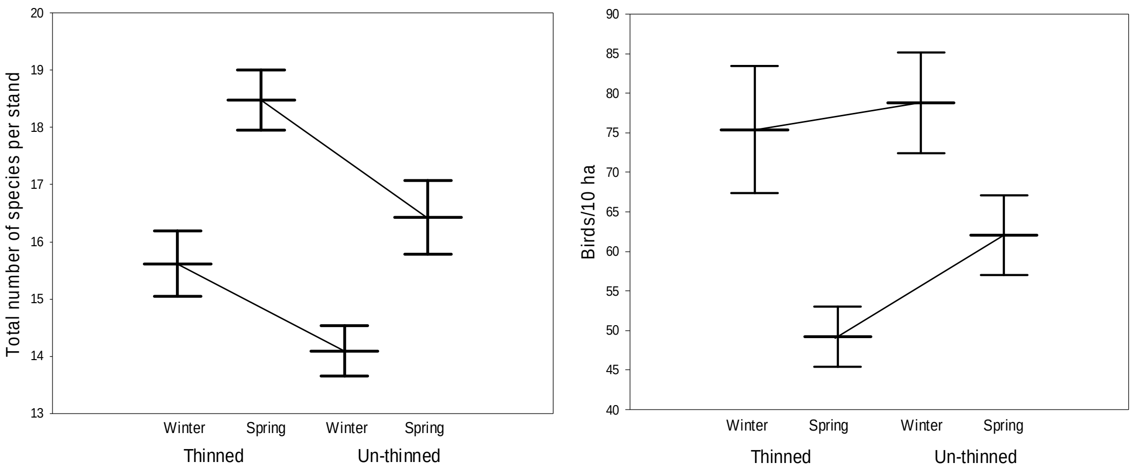


Figure 5.1. Variation in total bird species richness (left) and average density (rigth) in thinned and un-thinned (reference) stands across two study seasons. Bars represent mean ± one standard error. The census plots cover an area of 0.78 ha (i.e., circular plots of 50 m radius). Sample size is 21 stands.

Table 5.2. Results of the repeated measures ANOVAs testing for the effects of thinning and seasonal differences (winter vs. spring) in census plots. In the analyses of guilds, the effect of thinning on winter densities of birds is tested using the sample of winter censuses (i.e. no estimation of the effects of season and interaction terms). The analysis of endangered status is performed with data during the breeding season, because SPEC scores refer to European breeding populations. Degrees of freedom are 1, 20 for all tests.

	Thinning		Season		Interaction	
	F	*P*	*F*	*P*	*F*	*P*
a) Community variables						
Species richness	11.93	0.002	16.18	<0.001	0.37	0.550
Total density	2.65	0.119	16.14	<0.001	1.52	0.232
Average body mass	5.26	0.033	13.76	0.001	0.11	0.748
b) Foraging guilds						
Density of ground searchers	1.44	0.244	7.64	0.012	2.25	0.149
Density of foliage gleaners	10.14	0.005	20.44	<0.001	0.05	0.829
Winter density of granivorous birds	3.12	0.092				
Winter density of insectivorous birds	3.93	0.061				
Winter density of frugivorous birds	3.64	0.071				
c) European endangered status in spring	5.03	0.036				
d) Winter density of game birds	4.33	0.050				

spring, and chaffinch *Fringilla coelebs* L., woodpigeon *Columba palumbus*, woodlark *Lullula arborea*, magpie *Pica pica* L., azure-winged magpie *Cyanopica cooki*, mistle thrush *Turdus viscivorus*, and spotless starling *Sturnus unicolor* Temminck in winter. Conversely, only six common species were more abundant (> 33%) in un-thinned maquis than in thinned stands: sardinian warbler *Sylvia melanocephala* Gmelin, long-tailed tit *Aegithalos caudatus* L. and blackbird *Turdus merula* L. in both seasons; linnet *Carduelis cannabina* L. in spring; and dartford warbler *Sylvia undata* and firecrest *Regulus ignicapillus* Temminck in winter.

Thinning was responsible for a significant increase in species richness, but did not have any effect on bird density. Species richness and bird density significantly changed between seasons, richness being higher in spring than in winter, and density being higher in winter than in spring (Fig. 5.1, Table 5.2a). The interaction term season x thinning was never significant; thus, the effects of thinning were consistent and homogenous across seasons.

Average body mass of bird species in thinned stands was significantly larger than in un-thinned maquis, and in spring than in winter (Fig. 5.2, Table 5.2a). The interaction term season x thinning was not significant; thus, the effect of thinning on average body mass of bird was consistent and homogenous across seasons.

Effects of thinning on guilds

Density of ground searchers was indistinguishable in thinned and in un-thinned stands, whereas density of foliage gleaners was higher in un-thinned than in thinned stands. These two foraging guilds were denser in winter than in spring (Tables 5.2b, 5.3a). These patterns were consistent across seasons and thinning treatments (the interaction term thinning x season was not significant).

Winter density of granivorous species was marginally higher in thinned stands, whereas insectivorous and frugivorous species were marginally more abundant in un-thinned maquis (Tables 5.2b, 5.3a).

Effects of thinning on conservation status and density of gamebirds

The average SPEC score of the bird species was significantly different in thinned than in un-thinned stands during the breeding season (Tables 5.2c, 5.3b). Thinned areas support greater densities of bird species of high conservation status.

Winter density of game birds was significantly affected by thinning, being higher in thinned than in un-thinned stands (Tables 5.2d, 5.3c).

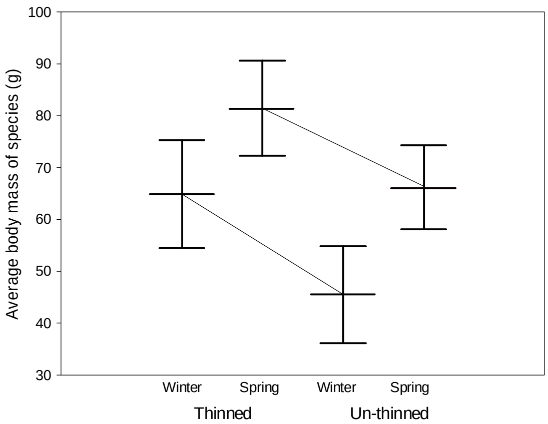


Figure 5.2. Variation in average body mass of bird species in thinned and un-thinned (reference) stands across two study seasons. Bars represent mean ± one standard error. Sample size is 21 stands. The average body mass of bird species in each census plots was calculated by means of weighted averages, using the species' densities in each locality stand as weights.

Discussion

Thinning and vegetation structure

Thinning eliminates the understorey woody vegetation to reduce competition for the biggest oaks resulting in a more open woodland that favours the growth of the herbaceous layer. It has the potential to increase the structural diversity of unmanaged monotonous stands, promoting the development of habitat attributes characteristic of later-serial habitats (Hagar, Howlin & Ganio 2004). Our data indicate that, 4 years on average after thinning, the herbaceous, short shrub, and tree cover increased by 50-100%. These changes are due to increased light availability, regrowth, and reduced intra- and inter-individual competition between trees (Perevolotsky & Haimov 1992; Haveri & Carey 2000; Artman 2003). On the other hand, the cover of tall shrubs (> 100 cm) and saplings (< 5 cm d.b.h.) was reduced by 100-200% (Table 5.1). We estimate that in ca. 12 years, under similar climate conditions, the thinned stands would attain the previous vegetation density. This represents a relatively rapid recovery of vegetation after disturbance. Consequently, thinned stands need to be maintained through further periodic thinning or grazing by cattle, and other natural disturbances such as fire would maintain open stands (Perevolotsky & Haimov 1992; Kaplan & Gutman 1996; Sekercioglu 2002; Brotons *et al.* 2003; Brotons, Herrando & Martin 2004).

Thinning and bird communities

Thinning was responsible for a significant increase in bird species richness. The homogeneous habitat structure of the natural maquis in the study area supports a poor bird community dominated by a low number of

Table 5.3. Mean and standard error (SE) of variables describing the bird communities in paired thinned and un-thinned (reference) stands during the winter and breeding (spring) seasons. *n* = 21 localities. Figures for densities are averages in five 50-m radius plots (i.e., birds 0.78 ha^{-1}).

		Thinned		Reference	
		Winter	Spring	Winter	Spring
a) Foraging guilds					
Density of ground searchers	Mean	2.60	1.58	2.14	1.62
	SE	0.34	0.18	0.26	0.21
Density of foliage gleaners	Mean	3.25	1.94	4.01	2.83
	SE	0.39	0.20	0.45	0.26
Winter density of granivorous birds	Mean	1.56	-	0.97	-
	SE	0.36	-	0.14	-
Winter density of insectivorous birds	Mean	3.73	-	4.51	-
	SE	0.37	-	0.46	-
Winter density of frugivorous birds	Mean	0.47	-	0.70	-
	SE	0.12	-	0.17	-
b) European endangered status in spring	Mean	-	3.80	-	3.96
	SE	-	0.07	-	0.10
c) Winter density of game birds	Mean	0.82	-	0.48	-
	SE	0.21	-	0.12	-

common species such as the sardinian warbler, long-tailed tit, dartford warbler, chaffinch, great tit and blue tit. The structural diversity of thinned stands creates habitats suitable for several bird species that are scarce in dense maquis (e.g. corn bunting, woodchat shrike, thekla lark, bee-eater, woodlark, magpie, azure-winged magpie, mistle thrush, woodpigeon and spotless starling). By contrast, thinning decreased the density of several previously dominant species (e.g. sardinian warbler, dartford warbler, long-tailed tit, firecrest), but no species were lost from stands following this practice. These results are consistent with those obtained from other studies of bird responses to thinning in different regions across the northern hemisphere (Haveri & Carey 2000; Artman 2003; Thompson, Baker & Ter-Mikaelian 2003; Hagar, Howlin & Ganio 2004). The higher species richness in thinned stands could be explained by the intermediate disturbance hypothesis, which predicts a peak of diversity under a moderate removal of biomass compared to undisturbed natural habitats (Solonen 1996; Wilkinson 1999). In our study, alteration of the homogenous structure of maquis by thinning led to an increase in habitat heterogeneity which is the main factor responsible for diversity in bird communities (see Wiens 1989 and references therein). The effects of thinning on Mediterranean bird communities appear to mirror those induced by other disturbances such as fire (Herrando, Brotons & Llacuna 2003; Brotons, Herrando & Martin 2004). Habitat heterogeneity at a local scale seems to be a key factor in maintaining bird diversity in disturbed Mediterranean landscapes. In particular, open-space species responded positively to changes in habitat structure leading to increased local abundance. Thus, it appears that changes in the structure of vegetation due to different ecological disturbances, such as fire and thinning, may promote similar changes on these bird communities.

The significance of the effects on the birds of this region was generally low, in spite of the marked changes in habitat structure introduced by maquis thinning. This contrasts with the observed patterns in other woodland environments of northern latitudes but may be explained, at least partially, by considering the biogeographic basis of the avifauna in the Mediterranean region. Mediterranean bird species are restricted to the early stages of succession and are then replaced by temperate forest species (Preiss, Martin & Debussche 1997). Xeric woodland environments of this region support an impoverished European forestal avifauna (Mönkkönen 1994; Covas & Blondel 1998). Density and species richness decrease towards the south of the Iberian Peninsula, and follow a pattern that is inversely related to rainfall and directly associated with temperature (Tellería & Santos 1993), the main determinants of primary productivity in dry Mediterranean environments (Mooney & Kummerow 1981). Only generalist woodland species (such as the chaffinch and the great tit) and those species whose distribution centres are located in the Mediterranean basin inhabit these woodland stands (e.g. dartford and black-headed warblers). Therefore, the low impacts of thinning on the avifauna of the study region can be attributed to the small regional pool of species typical of well developed woodlands, and the fact that the dominant bird species in this region are those mainly associated with shrublands and evergreen forests of low tree height (Suárez-Seoane, Osborne & Baudry 2002).

A dense undergrowth of shrubs and young trees inhibits the development of a complex shrub and herbaceous strata which therefore limits the feeding opportunities for ground and shrub gleaners (for similar results in other forest types see Haveri & Carey 2000; Hayes, Weikel & Huso 2003; Hagar, Howlin & Ganio 2004). The elimination of this dense understorey through thinning operations would be expected to increase the abundance of these functional guilds and limit the abundance of foliage gleaners and insectivorous birds. Both predictions are supported by our results, which have been also reported in other forest environments (Easton & Martin 1998).

Thinning and body size of bird species

Slender and pliable substrata, such as foliage and thin twigs, are mainly exploited by small-sized birds because of eco-morphological constraints on manoeuvrability (Miles & Ricklefs 1984; Suhonen, Alatalo & Gustafsson 1994). Accordingly, the average body mass of bird assemblages is related to habitat structure and the use of foraging substrata (Gunnarsson 1992; Dixon, Kindlmann & Jarosik 1995; Polo & Carrascal 1999). Bird species that thrive in densely vegetated habitats and forage on foliage are lighter than those living in open habitats that forage on the ground (Polo & Carrascal 1999). Our results are in agreement with these patterns, as the average body mass of species in thinned stands was significantly higher than in un-thinned maquis. To our knowledge, this is the first time that a large-scale experimental manipulation of habitat structure and vegetation volume has demonstrated the predicted allometric effect of habitat structural complexity on the average body mass of a bird community. This relationship had been previously observed from comparisons of several species differing in body mass and habitat preferences. Nevertheless, these traits have a remarkable phylogenetic correlation (Polo & Carrascal 1999; Freckleton, Harvey & Pagel 2002), so the relationship between them may emerge as a consequence of the evolutionary history of the group. In ecological time, the results of our study can be explained if we assume that structural complexity and vegetation density act as a selective filter of the bird fauna of the region, allowing the establishment and promoting the increase of density of small-sized species in the densely vegetated, un-thinned maquis. The higher average body mass in thinned stands is a consequence of both the increased density of relatively large species that mainly forage on the ground and the decreased density of relatively small species that usually forage on the foliage and twigs affected by thinning.

Implications for species conservation and hunting

We found that thinned stands supported bird species of high conservation concern in Europe. We did not record any species threatened with extinction (SPEC 1). Species such as red-legged partridge, woodchat shrike and wood lark, with SPEC 2, tended to be more abundant in thinned than un-thinned stands, making this habitat important for species conservation. These results have to be considered cautiously since these species are actually rather common at the regional level. Nevertheless, we point out that thinning did not adversely affect the local abundance of endangered species in our study. Since thinned and un-thinned stands favoured different guilds, management directed toward a single species or species group and extensive, conventional forestry are unlikely to be successful for the conservation of bird communities that inhabit complex forest ecosystems (Artman 2003; Carey 2003; Thompson, Baker & Ter-Mikaelian 2003; Hagar, Howlin & Ganio 2004). Both thinned and un-thinned areas should be maintained.

Hunting is an important economic activity in the region, and provides more monetary value per area unit than, for example, agriculture. The value of forest management for hunting has been reported for several game species (e.g. Terry, McLellan & Watts 2000). The winter abundance of game birds increased in thinned stands because these birds are mostly granivorous and frugivorous species that prefer to forage in this habitat rather than in dense maquis (for wintering birds in Central Spain see Carrascal, Palomino & Lobo 2002). However, they rely on more dense vegetation patches (red-legged partridge,) and trees (wood pigeon, turtle dove, song thrush and mistle thrush) for nesting. Thus, a habitat mosaic of dense and open woody vegetation (Scarascia-Mugnozza *et al.* 2000) is necessary for the maintenance of populations of valuable game birds.

In conclusion, our results indicate that thinning of the Mediterranean maquis modifies the structure of the vegetation and, as a consequence, also alters several aspects of bird community structure, including species richness, guild composition and average body mass. The effects were consistent across seasons except for body mass. Thinning of dense Mediterranean woodlands dominated by shrubs and young trees enhances habitat suitability for several species of birds, but some un-thinned patches should be preserved to provide refugia for species that are impacted by thinning (Hayes, Weikel & Huso 2003; Hagar, Howlin & Ganio 2004). Silvicultural thinning was also found to be beneficial for threatened bird species in the European context and for game birds, emphasizing both its conservation and economical values.

Acknowledgments

This study has been funded by the grant projects REN2000-745 and CGL2004-355-BOS (CICYT, Ministry of Education and Science, Spain). It received additional input from project GR/AMB/0757/2004 and the REMEDINAL Research Network (S-0505/AMB/0335) (Madrid Autonomous Community). EM received research fellowships from the University of Alcala and the Madrid Autonomous Community. We acknowledge the permission from various stakeholders to survey their properties. We are indebted to Salvador Rebollo for his preliminary comments on field sampling. Claire Jasinski kindly improved the English of the manuscript. Two anonymous reviewers provided useful comments and suggestions that improved a previous version of this manuscript.

References

Artman, V.L. (2003) Effects of commercial thinning on breeding bird populations in western hemlock forests. American Midland Naturalist, 149, 225-232.

Bibby, C.J., Burgess, N.D., Hill, D.A. & Mustoe, S.H. (2000) Bird census techniques. Academic Press, London.

Blondel, J. & Aronson, J. (1999) Biology and wildlife of the Mediterranean region. Oxford University Press, New York.

Brotons, L., Mönkkönen, M., Huhta, E., Nikula, A. & Rajasärkkä, A. (2003) Effects of landscape structure and forest reserve location on old-growth forest bird species in Northern Finland. Landscape Ecology, 18, 377-393.

Brotons, L., Herrando, S. & Martin, J.L. (2004) Bird assemblages in forest fragments within Mediterranean mosaics created by wild fires. Landscape Ecology, 19, 663-675.

Carey, A.B. (2003) Biocomplexity and restoration of biodiversity in temperate coniferous forest: inducing spatial heterogeneity with variable-density thinning. Forestry, 76, 127-136.

Carrascal, L.M., Palomino, D. & Lobo, J.M. (2002) Patterns of habitat preference and distribution and abundance of wintering bird fauna in Central Spain. Analisys and prediction of the effect of ecological factors. Animal Biodiversity and Conservation, 25, 7-40.

Covas, R. & Blondel, J. (1998) Biogeography and history of the Mediterranean bird fauna. Ibis, 140, 395-407.

DeBell, D.S., Curtis, R.O., Harrington, C.A., & Tappeiner, J.C. (1997) Shaping stand development through silvicultural practices. Creating a Forest for the 21st Century: The Sciences of Ecosystem Management (eds. K.A. Kohm & J.F. Franklin), pp. 141-149, Island Press, Washington, DC.

Díaz, M., Carbonell, R., Santos, T. & Tellería, J.L. (1998) Breeding bird communities in pine plantations of the Spanish plateaux: biogeography, landscape and vegetation effects. Journal of Applied Ecology, 35, 562-574.

Dixon, A.F.G., Kindlmann, P. & Jarosik, V. (1995) Body-size distribution in aphids - relative surface-area of specific plant structures. Ecological Entomology, 20, 111-117.

Ducrey, M. & Toth, J. (1992) Effect of cleaning and thinning on height growth and girth increment in holm oak coppices (*Quercus ilex*). Vegetatio, 100, 365-376.

Easton, W.E. & Martin, K. (1998) The effect of vegetation management on breeding bird communities in British Columbia. Ecological Applications, 8, 1092-1103.

Freckleton, R.P., Harvey, P.H. & Pagel, M. (2002) Phylogenetic analysis and comparative data: A test and review of evidence. American Naturalist, 160, 712-726.

Gaston, K.J. & Blackburn, T.M. (2000) Pattern and process in macroecology. Blackwell Science, Oxford.

Grove, A.T. & Rackham, O. (2001) The nature of Mediterranean Europe. An ecological history. Yale University Press, New Haven.

Gunnarsson, B. (1992) Fractal dimension of plants and body size distribution in spiders. Functional Ecology, 6, 636-641.

Hagar, J., Howlin, S. & Ganio, L. (2004) Short-term response of songbirds to experimental thinning of young Douglas-fir forests in the Oregon Cascades. Forest Ecology and Management, 199, 333-347.

Halaj, J., Ross, D.W. & Moldenke, A.R. (2000) Importance of habitat structure to the arthropod food-web in Douglas-fir canopies. Oikos, 90, 139-152.

Haveri, B.A. & Carey, A.B. (2000) Forest management strategy, spatial heterogeneity, and winter birds in Washington Source: Wildlife Society Bulletin, 28, 643-652.

Hayes, J.P., Weikel, J.M. & Huso, M.M.P. (2003) Response of birds to thinning young Douglas-fir forests. Ecological Applications, 13, 1222-1232.

Heikkinen, R.K., Luoto, M., Virkkala, R. & Rainio, K. (2004) Effects of habitat cover, landscape structure and spatial variables on the abundance of birds in an agricultural-forest mosaic. Journal of Applied Ecology, 41, 824-835.

Herrando, S., Broton, L. & Llacuna, S. (2003) Does fire increase the spatial heterogeneity of bird communities in Mediterranean landscapes? Ibis, 145, 307-317.

Homyack, J.A., Harrison, D.J. & Krohn, W.B. (2004) Structural differences between precommercially thinned and unthinned conifer stands. Forest Ecology and Management, 194, 131-143.

Kaplan, D. & Gutman, M. (1996) Effect of thinning and grazing on tree development and the visual aspect of an oak forest on the Golan Heights. Israel Journal of Plant Sciences, 44, 381-386.

Liow, L.H., Sodhi, N.S. & Elmqvist, T. (2001) Bee diversity a along disturbance gradient in tropical lowland forests of south-east Asia. Journal of Applied Ecology, 38, 180-192.

Maldonado, J., Benito, M., Sanchez, R.A. & Sainz, H. (2002) Evolución reciente de las áreas de los bosques esclerófilos ibéricos. Cambios deducidos a partir de la cartografía forestal. La regeneración natural del bosque Mediterráneo en la Península Ibérica. (eds. J. Charco), pp. 217-236, ARBA-Ministerio de Medio Ambiente, Madrid.

Marañón, T., Ajbilou, R., Ojeda, F. & Arroyo, J. (1999) Biodiversity of woody species in oak woodlands of southern Spain and Northen Morocco. Forest Ecology and Management, 115, 147-156.

Miles, D.B. & Ricklefs, R.E. (1984) The correlation between ecology and morphology in deciduous forest passerine birds. Ecology, 65, 1629-1640.

Mönkkönen, M. (1994) Diversity patterns in Paleartic and Neartic forest bird assemblages. Journal of Biogeography, 21, 183-195.

Mooney, H.A. & Kummerow, J. (1981) Phenological development of plants in Mediterranean-climate regions. Ecosystems of the World II. Mediterranean type shrublands (eds. F.D. Castri, D.W. Goodall, R.L. Specht), pp. 303-307, Elsevier, Amsterdam.

Patriquin, K.J. & Barclay, R.M.R. (2003) Foraging by bats in cleared, thinned and unharvested boreal forest. Journal of Applied Ecology, 40, 646-657.

Perrins, C. (1998) The complete birds of the Western Palearctic on CD-ROM. Oxford University Press, Oxford.

Perevolotsky, A. & Haimov, Y. (1992) The effect of thinning and goat browsing on the structure and development of Mediterranean woodland in Israel. Forest Ecology and Management, 49, 61-74.

Polo, V. & Carrascal, L.M. (1999) Shaping the body size distribution of passeriformes: habitat use and body size are evolutionarily and ecologically related. Journal of Animal Ecology, 68, 324-337.

Preiss, E.C., Martin, J.L. & Debussche, M. (1997) Rural depopulation and recent changes in a Mediterranean region: consequences to the breeding avifauna. Landscape Ecology, 12, 51-61.

Sánchez-Zapata, J.A. & Calvo, J.F. (1999) Raptor distribution in relation to landscape composition in semi-arid Mediterranean habitats. Journal of Applied Ecology, 36, 245-262.

Scarascia-Mugnozza, G., Oswald, H., Piussi, P. & Kalliopi, R. (2000) Forest of the Mediterranean region: gaps in knowledge and research needs. Forest Ecology and Management, 132, 97-109.

Sekercioglu, C.H. (2002) Effects of forestry practices on vegetation structure and bird community of Kibale National Park, Uganda. Biological Conservation, 107, 229-240.

Shiu, H.J. & Lee, P.F. (2003) Assessing avian point-count duration and sample size using species accumulation functions. Zoological Studies, 42, 357-367.

Solonen, T. (1996) Patterns and variations in the structure of forest bird communities in southern Finland. Ornis Fennica, 73, 12-26.

StatSoft (2001) STATISTICA (data analysis software system), version 6., StatSoft, Inc., Tulsa, OK, USA.

Suárez-Seoane, S., Osborne, P.E. & Baudry, J. (2002) Responses of birds of different biogeographic origins and habitat requirements to agricultural land abandonment in northern Spain. Biological Conservation, 105, 333-344.

Suhonen, J., Alatalo, R.V. & Gustafsson, L. (1994) Evolution of foraging ecology in Fennoscandian tits (*Parus* spp). Proceedings of the Royal Society of London, 258, 127-131.

Sullivan, T.P., Lautenschlager, R.A. & Wagner, R.G. (1999) Clearcutting and burning of northern spruce-fir forests: implications for small mammal communities. Journal of Applied Ecology, 36, 327-344.

Sullivan, T.P., Sullivan, D.S. & Lindgren, P.M.F. (2001) Stand structure and small mammals in young lodgepole pine forest: 10-year results after thinning

Ecological Applications, 11, 1151-1173.

Tellería, J.L. & Santos, T. (1993) Distributional patterns of insectivorous passerines in the Iberian forests: does abundance decrease near the border? Journal of Biogeography, 20, 235-240.

Terry, E.L., McLellan, B.N. & Watts, G.S. (2000) Winter habitat ecology of mountain caribou in relation to forest management. Journal of Applied Ecology, 37, 589-602.

Thomas, S.C., Halpen, C.B., & Falk, D.A. (1999) Plant diversity in managed forests: Understory responses to thinning and fertilization. Ecological Applications, 9, 864-879.

Thompson, I.D., Baker, J.A. & Ter-Mikaelian, M. (2003) A review of the long-term effects of post-harvest silviculture on vertebrate wildlife, and predictive models, with an emphasis on boreal forests in Ontario, Canada. Forest Ecology and Management, 177, 441-469.

Tucker, G.M. & Heath, M.F. (1994) Birds in Europe: their conservation status. BirdLife International, Cambridge.

Wiens, J. (1989) The ecology of bird community. Vol I: Foundations and patterns. Cambridge University Press, Cambridge.

Wilkinson, D.M. (1999) The disturbing history of intermediate disturbance. Oikos, 84, 145-147.

Wainwright, J., Parsons, A.J. 2002. The effect of temporal variations in rainfall on scale dependency in runoff coefficients. Water Resources Research, 38: 1271. DOI: 10.1029/2000WR000188.

Whisenant, S.G., 2002. Manipulation of the physical environment. Terrestrial systems. In: Perrow, M.R., Davy, A.J. (Eds.), Handbook of Ecological Restoration. Volume1. Principles of Restoration. Cambridge University Press, Cambridge, UK. Pp.: 83-105.

Wilcox, B.P., Newman, B.D. 2005. Ecohydrology of semiarid landscapes. Ecology, 86: 275-276.

Wilcox, B.P., Pitlick, J., Allen, C.D., Davenport, D.W. 1996. Runoff and Erosion from a rapidly eroding pinyon-juniper hillslope. In: Anderson, M.G., Brooks, S.M. (Eds.), Advances in Hillslope Processes, John Wiley and Sons, Chichester, UK, pp.: 61-77.

Wilcox, B.P., Breshears, D.D., Allen, C.D. 2003. Ecohydrology of a resource-conserving semiarid woodland: effects of scale and disturbance. Ecological Monographs, 73: 223-239.

Wischmeier, W.H., Smith, D.D. 1978. Predicting rainfall erosion losses: a guide to conservation planning. Agriculture Handbook 537. United Sates Department of Agriculture, Science and Education Administration. Washington, D.C., USA.

Yair, A., Kossovsky, A. 2002. Climate and surface properties: hydrological response of small arid and semi-arid watersheds. Geomorphology, 42: 43-57.

Yair, A., Lavee, H. 1985. Runoff generation in arid and semi-arid areas. In: Anderson, M.G., Burt, T.P. (Eds.), Hydrological Forecasting. John Wiley and Sons Ltd., Chichester, UK. Pp.: 183-220.

Yair, A., Raz-Yassif, N. 2004. Hydrological processes in a small arid catchment: scale effects of rainfall and slope length. Geomorphology, 61: 155-169.

Appendix 5.A: Densities (birds/10 ha) of species in the thinned and un-thinned (reference) stands of the study area in Ciudad Real (Central Spain). One hundred and five circular census plots (50 m in radius) were surveyed in 21 localities in both thinned and reference stands. Each census plot was repeated two times per year, in two consecutive years (i.e. four replicates per census plot). ---: species absent. SPEC scores define the European conservation status: 5 (non-endangered category), 4, 3, 2 (the most endangered category). Winter foraging guilds: carnivorous (C), granivorous (G), insectivorous (I), frugivorous (F). Main foraging substrata: aerial feeder (a), foliage gleaners (f), ground searchers (g), trunk foragers (t). Game species (H).

		Thinned stands		Reference stands	
		Winter	Spring	Winter	Spring
Long-tailed tit	*Aegithalus caudatus* (5,I,f)	5.43	2.20	12.76	7.51
Red-legged partridge	*Alectoris rufa* (2,G,g,H)	2.93	2.20	3.11	1.77
Buzzard	*Buteo buteo* (5,C,g)	0.06	0.06	---	---
Linnet	*Carduelis cannabina* (4,G,g)	0.37	0.24	0.12	1.89
Goldfinch	*Carduelis carduelis* (5,G,g)	0.43	0.31	0.37	0.55
Greenfinch	*Carduelis chloris* (4,G,g)	---	0.12	---	0.37
Siskin	*Carduelis spinus* (4,G,f)	0.24	---	---	---
Short-toed treecreeper	*Certhia brachydactyla* (4,I,t)	---	0.12	0.37	0.37
Great spotted cuckoo	*Clamator glandarius* (5,g)	---	0.37	---	0.31
Woodpigeon	*Columba palumbus* (4,G,g,H)	2.08	2.32	0.73	2.20
Quail	*Coturnix coturnix* (3,g,H)	---	0.12	---	---
Cuckoo	*Cuculus canorus* (5,g)	---	0.12	---	0.06
Azure-winged magpie	*Cyanopica cyana* (4,I,g)	1.59	3.11	0.31	2.50
Rock bunting	*Emberiza cia* (3,G,g)	0.31	0.24	0.43	0.12
Cirl bunting	*Emberiza cirlus* (4,G,g)	---	---	---	0.12
Robin	*Erithacus rubecula* (4,I,g)	12.51	0.12	14.71	---
Chaffinch	*Fringilla coelebs* (4,G,f)	9.46	5.43	5.49	4.88
Thekla lark	*Galerida theklae* (3,G,g)	0.31	1.59	0.55	0.98
Jay	*Garrulus glandarius* (5,G,g/f)	---	0.31	---	0.24
Melodious warbler	*Hippolais polyglotta* (4,f)	---	0.06	---	0.12
Swallow	*Hirundo rustica* (3,a)	---	0.85	---	0.24
Great grey shrike	*Lanius meridionalis* (3,I,g)	---	0.31	---	---
Woodchat shrike	*Lanius senator* (2,g)	---	1.53	---	0.49
Wood lark	*Lullula arborea* (2,G,g)	1.65	0.49	0.49	0.18
Bee-eater	*Merops apiaster* (3,a)	---	2.50	---	1.22
Corn bunting	*Miliaria calandra* (4,G,g)	---	1.40	0.31	0.37
Spotted flycatcher	*Muscicapa striata* (3,a)	---	---	---	0.31
Blue tit	*Parus caeruleus* (4,I,f)	4.09	2.32	5.25	2.93
Crested tit	*Parus cristatus* (4,I,f)	0.18	0.24	0.31	0.06
Great tit	*Parus major* (5,I,f)	4.70	3.24	5.37	3.11
Spanish sparrow	*Passer hispaniolensis* (5,G,g)	---	0.43	---	0.12
Tree sparrow	*Passer montanus* (5,G,g)	---	---	---	0.18
Black redstart	*Phoenicurus ochruros* (5,I,g)	0.24	---	---	---
Bonelli's warbler	*Phylloscopus bonelli* (4,f)	0.18	0.43	---	0.43
Chiffchaff	*Phylloscopus collybita* (5,I,f)	4.94	---	5.19	---
Magpie	*Pica pica* (5,C,g)	1.59	0.18	0.06	0.73
Greenn woodpecker	*Picus viridis* (2,I,g/t)	0.06	0.06	0.06	0.24
Dunnock	*Prunella modularis* (4,G,g)	0.49	---	0.49	0.06
Crag martin	*Ptyonoprogne rupestris* (5,I,a)	0.92	0.31	---	0.92
Firecrest	*Regulus ignicapillus* (4,I,f)	2.26	0.37	4.46	---
Serin	*Serinus serinus* (4,G,g)	0.49	1.65	0.37	2.14
Turtle dove	*Streptopelia turtur* (3,g,H)	---	0.12	---	0.43
Spotless starling	*Sturnus unicolor* (4,G,g)	1.22	---	---	0.24
Blackcap	*Sylvia atricapilla* (4,F,f)	0.06	0.18	---	0.18
Subalpine warbler	*Sylvia cantillans* (4,f)	0.18	0.73	---	1.04
Orphean warbler	*Sylvia hortensis* (3,f)	---	0.18	---	0.06
Sardinian warbler	*Sylvia melanocephala* (4,F,f)	2.99	5.92	5.49	10.13
Dartford warbler	*Sylvia undata* (2,I,f)	6.90	3.36	7.14	5.68
Blackbird	*Turdus merula* (5,F,g)	1.40	2.26	3.11	3.97
Song thrush	*Turdus philomelos* (4,I,g)	3.97	---	1.89	---
Mistle thrush	*Turdus viscivorus* (4,F,g)	1.59	0.37	0.37	0.55
Hoopoe	*Upupa epops* (5,I,g)	0.06	0.24	---	0.12
Total density		75.88	48.72	79.30	60.13
Number of species		34	43	27	43
Diversity (Shannon index)		2.87	3.13	2.59	2.96

Capítulo 6

"Austeridad, no pobreza (...) la tierra tiene suficientes recursos para cubrir las necesidades de todos, aunque no para la codicia de algunos"

Mahatma Gandhi

La sesión del Chullachaki. El espíritu de la selva transmite instrucciones para el cuidado de los animales salvajes. Pablo Amaringo, Chamán peruano.

Capítulo 6

Consideraciones adicionales

Como biólogo puedo asegurar que el trabajar en y para la Naturaleza es suficientemente gratificante como para mantener radiante mi parcela de felicidad profesional. Pero me gusta pensar que el fin último de todos los que dedicamos nuestros esfuerzos en pro de la conservación de la Naturaleza es el bienestar de la humanidad, además de por respeto a la vida en general. Nuestra incesante transformación y degradación de los ecosistemas está dañando el bienestar humano global a cambio de unos beneficios a corto plazo. En estas circunstancias, realizar una razonable combinación de conservación y explotación sostenible de los ecosistemas tiene un sentido tanto moral como económico.

Diversidad y conservación

Nos encontramos en una fase crucial en el desarrollo del conocimiento teórico y su aplicación a las estrategias de conservación. Por ello hemos centrado este trabajo en generar un conjunto de herramientas e información de interés para la conservación de la diversidad biológica. Como ejemplo, podemos destacar el Índice Combinado de Biodiversidad que hemos desarrollado y evaluado en esta Tesis. Debido a la falta de congruencia encontrada entre las distintas medidas de diversidad empleadas comúnmente para identificar áreas relevantes de diversidad, el uso del Índice Combinado evitaría la potencial ineficiencia que se produce al usar una sola de las medidas que integra, ya que además de incorporar la riqueza, la rareza y la vulnerabilidad de las especies en una única medida, es más eficiente, como hemos podido comprobar en los capítulos 2, 3 y 4.

La identificación de áreas relevantes de diversidad es considerada una estrategia útil y rentable en términos de conservación (Myers *et al*. 2000). La metodología propuesta puede incluirse dentro de la Planificación Sistemática de la Conservación (Margules & Pressey 2000), y también es de utilidad para la priorización de otras actuaciones como la mitigación de impactos ambientales y la restauración ecológica, además de como base para futuras investigaciones. No obstante existen problemas relativos a la selección de áreas, los datos de base, la escala de análisis, ciertas variables socioeconómicas y el mantenimiento de procesos clave que pueden provocar que se excluyan de las estrategias de conservación áreas con alto valor ecológico (ver Whittaker *et al*. 2005 para una revisión del tema).

En los estudios macroecológicos el problema más frecuente es la falta de datos fiables sobre la distribución de grupos de especies que no están bien definidos taxonómicamente, así como por el inadecuado conocimiento de la distribución de muchas especies debido a la ausencia o escasez de trabajos de investigación, los llamados *Linnean shortfall* (Brown & Lomolino 1998) y *Wallacean shortfall* (Lomolino 2004), respectivamente. El distinto esfuerzo de muestreo realizado en diferentes áreas geográficas y taxones produce sesgos en los datos de base que condicionan los resultados finales. Por otro lado, los análisis deberían realizarse con datos independientes para varios taxones, ya que las prioridades identificadas para un taxón pueden no reflejar adecuadamente la diversidad de otros taxones (Van Jaarsveld *et al*. 1998). Lo ideal sería que los análisis incluyeran a los invertebrados y las plantas, ya que representan la mayor parte de la biodiversidad. Pero la realidad es que se dedica la mayor parte de los recursos y esfuerzo a los vertebrados.

La escala espacial de análisis es un factor determinante de los patrones de diversidad, debiéndose distinguir dos propiedades distintas: la extensión geográfica del sistema de estudio y el grano o resolución de los datos. Existen trabajos que muestran distintos patrones de diversidad al variar la resolución de los datos mientras se mantiene constante la extensión geográfica, y viceversa (Rahbek & Graves 2000, Koleff & Gaston 2002, Hawkins *et al*. 2003, Rahbek 2005, Hurlbert & Jetz 2007). Esto indica que algunos criterios de selección de áreas relevantes basados en datos de diversidad y distribución de especies pueden ser sensibles a la escala de análisis empleada. Aunque no se han hecho muchos trabajos para evaluar este hecho, parece que podría tener una fuerte influencia (Araujo 2004), y por lo tanto son necesarias nuevas técnicas que permitan análisis útiles a diferentes escalas.

Muchas veces las áreas relevantes de diversidad establecidas son demasiado amplias para dirigir las actividades de conservación, gestión y restauración en escalas realistas de trabajo. Establecer áreas del territorio para prioridades de conservación basada en la biodiversidad puede ignorar muchos factores sociales relevantes. Si se pretende generar información útil para la correcta asignación de los recursos destinados a la conservación, se deben incluir factores económicos y sociales (Moore *et al*. 2004, Fraga 2006). También es conveniente que los patrones espaciales de biodiversidad sean analizados a escala local con el objetivo de facilitar una conexión directa entre conocimiento biológico y las intervenciones concretas de conservación (Harris *et al*. 2005). Es necesario que las actuaciones en áreas reducidas como los resalveos, u otras encaminadas a la conservación o el aprovechamiento de los recursos naturales, se integren dentro de estrategias de conservación más amplias, que sean el fruto de la coordinación entre organismos y/o administraciones, y así poder evitar inconsistencia en la gestión del territorio (Failing & Gregory 2003). A escala de proyecto, es necesario disponer del conocimiento científico básico para poder adaptar la gestión a las peculiaridades ecológicas y socioeconómicas locales, mientras se cumplen los objetivos de conservación a escala de planificación.

La distribución de la riqueza de especies, especies amenazadas y especies endémicas es determinada por la interacción de distintos factores biológicos, ecológicos, evolutivos y humanos. Es necesario un mayor conocimiento teórico que explique los diferentes patrones de diversidad, aunque hay que tener en cuenta que existen evidencias de que el mantenimiento de los patrones de distribución de especies y comunidades no garantiza por sí mismo la conservación de los procesos evolutivos y ecológicos clave (Balmford *et al*. 1998, Pressey *et al*.

2003). Es importante considerar la complementaridad entre las distintas unidades de un territorio, ya que ayuda a flexibilizar la selección de éstas y a maximizar la eficiencia en conservación y representación minimizando los costes socioeconómicos. También son necesarias aproximaciones que evalúen la probabilidad de persistencia de las poblaciones en el futuro, en particular, en los distintos escenarios de cambio climático que se plantean actualmente (IPCC 2007).

Conservación de la Naturaleza y sociedad

Considerando las limitaciones y la constante necesidad de mejorar el conocimiento alcanzado, la realidad es que, por ejemplo, mucha de la superficie protegida en el mundo es consecuencia de oportunidades locales más que el resultado de una evaluación *a priori* de sus valores ecológicos (Prendergast *et al*. 1999). Esto se produce a pesar de la existencia de multitud de métodos y técnicas condenados a ser siempre teóricamente útiles, y que jamás llegarán a demostrar su aplicación práctica. La Convención sobre la Diversidad Biológica reconoce que se cuenta con los instrumentos necesarios para alcanzar el objetivo de reducir la pérdida de biodiversidad para el año 2010 (UNEP 2002), pero que lo más complicado es lograr que sean utilizados por todos los sectores económicos, desde la pesca hasta la selvicultura, desde la agricultura hasta la industria, desde la planificación hasta el comercio. Después de varios años de buenas ideas y esfuerzos la mayor parte de ellos siguen siendo letra muerta, a pesar de que la incertidumbre sobre la no linealidad y la reversibilidad de los efectos de la pérdida de biodiversidad en los ecosistemas reclame medidas inmediatas para mitigar las consecuencias negativas en el futuro (Chapin *et al*. 2000).

Todo ello se debe en parte a que la conservación de la Naturaleza es una tarea compleja debido a la interacción de los procesos ecológicos con los de índole socioeconómica, a la participación activa o pasiva, voluntaria u obligada, favorable o contraria de multitud de sectores sociales y a la distribución de todos estos elementos en una escala geográfica amplia. Esto fomenta la aparición de diferentes conflictos de intereses económicos, políticos, sociales e incluso personales. La conservación es un uso que se enfrenta a otros usos.

También influye el que frecuentemente no se consigue difundir el resultado de las investigaciones más allá del ámbito científico, bien porque no es suficientemente atractivo para su aplicación real, bien porque no tenemos o no sabemos usar las vías de comunicación adecuadas. Evaluar el impacto de un artículo científico en la política y la gestión es siempre difícil de demostrar, pero parece que se presta escasa consideración a los trabajos publicados en nuestro campo de investigación. Afortunadamente, algunas señales indican que la ciencia está alcanzando los círculos políticos. Un ejemplo es la publicación del *Bowral Checklist* (Lindenmayer *et al*. 2008), donde se identifican 12 temas importantes que deben ser considerados en el desarrollo de estrategias para la conservación de paisajes. Unas semanas después de su publicación, apareció un resumen de sus conclusiones en el *Science for Environment Policy*, el servicio de noticias ambientales de la Unión Europea, distribuido a unos 6000 políticos y legisladores. En cualquier caso, siempre es imprescindible la voluntad política de actuación y, por desgracia, ésta suele faltar. Frecuentemente se toman medidas cuando un "mal ecológico" ya está hecho y la opinión pública clama.

Pensar en el futuro de la biodiversidad a escala global es desalentador (Sala *et al*. 2000). Es más práctico y reconfortante considerar las amenazas a la integridad de los ecosistemas como una oportunidad concreta de reducir el problema a unas dimensiones más manejables, a una situación más soportable. ¿De qué forma podemos abordar estas amenazas para ser más efectivos y eficientes en la conservación? Una de las mejores apuestas para lograrlo es la integración y la actuación sobre los diferentes aspectos ecológicos, sociales, políticos, organizativos y económicos que rodean a la conservación. Esta integración debe incluir tanto herramientas técnicas y científicas como una nueva manera de ver la realidad que trascienda el pensamiento académico tradicional que nos obliga a compartimentarla para poder comprenderla y cambiarla.

La Ecología tiene la obligación de proveer a la sociedad el conocimiento necesario para gestionar sabiamente la Tierra y sus recursos biológicos. Esto requerirá comprender suficientemente la Naturaleza y los impactos humanos en ella, y que comuniquemos este conocimiento al conjunto de la sociedad. ¿Por qué limitarnos a dialogar dentro de la comunidad interesada en la conservación de los ecosistemas si ésta interesa a todos los sectores de la economía? ¿Cómo podemos presentar el conocimiento científico de manera útil a las administraciones, empresas y la sociedad en general? Parte de las respuestas a estas preguntas radica en establecer indicadores de los servicios de la biodiversidad y los ecosistemas que sean rigurosos, constantes, ampliamente aceptados y fácilmente comprensibles (Balmford *et al*. 2005a, Hezri & Dovers 2006, Mace & Baillie 2007). Los científicos de la conservación tenemos mucho que aprender en este tema de los economistas. Uno de esos indicadores ecológicos, el *U.K. Wild Bird Index*, basado en las tendencias de las poblaciones de aves nidificantes comunes, ha sido adoptado por el gobierno de Reino Unido como un indicador de la calidad de vida y una medida de cómo están funcionando las políticas ambientales.

Además de modelos que describan cómo interactúan los componentes humanos, biológicos, físicos y químicos de los sistemas de la Tierra, es necesario dar respuestas a preguntas específicas aplicables a situaciones reales. El problema es identificar cuáles son las preguntas de interés y quién las establece. La investigación está en posición de ofrecer el asesoramiento que necesitan administraciones y empresas para tomar decisiones importantes. Pero si no son la opción más rentable pocas veces son consideradas. Normalmente las ganancias a corto plazo se priorizan antes que cualquier valor a largo plazo asociado a la conservación del sistema natural que genere ingresos sostenidos. Existe un compromiso entre los beneficios actuales y los costes futuros de la degradación ambiental, y entre el beneficio de unos pocos y el coste para muchos. La investigación necesita cuantificar estos compromisos. Son necesarios trabajos relacionando ecología y economía para cuantificar los costes inmediatos y a largo plazo y los beneficios de acciones alternativas (Tilman 2000). La causa subyacente de la degradación ambiental es frecuentemente económica, y por lo tanto la solución debe también incorporar principios económicos. Los servicios de los ecosistemas no están adecuadamente contemplados dentro de los mercados comerciales, por lo que frecuentemente tienen muy poco peso en las decisiones políticas. El imperativo global de proteger los servicios de los ecosistemas y de la biodiversidad debe convertirse políticamente en algo tan relevante como lo es ahora el crecimiento económico (Balmford *et al*. 2005b).

Valores humanos y conservación

Ese modelo económico es irresponsable y lo que llamamos desarrollo es insostenible. Las actuales trayectorias de desarrollo evidencian la ausencia de beneficios para la humanidad de la manera que deberían: la desigualdad de ingresos *per capita* aumenta en todo el mundo y la mayoría de los países no están en el camino de alcanzar los objetivos de Naciones Unidas para el desarrollo humano y la erradicación de la pobreza en el año 2015 (UNDP 2000). En 2008 casi se alcanzó la cifra de mil millones de personas que pasan hambre en el mundo, unos 40 millones más que el año anterior (FAO 2008). El logro de un desarrollo verdaderamente sostenible obliga a reconsiderar los actuales paradigmas económicos y a rechazar las soluciones a corto plazo que, a la larga, no llevan a ninguna parte.

Hay que cambiar los valores fundamentales de nuestra sociedad materialista. La responsabilidad personal y el uso eficiente de los recursos debe ser una prioridad para las sociedades modernas. Es necesario que se reduzca el nivel de consumo individual de energía, recursos y alimentos producidos en los niveles tróficos más altos. Es inconcebible que el deseable bienestar de toda la humanidad desemboque en el nivel de consumo actual de los ciudadanos de los países derrochadores. Cada ser humano utiliza tanta energía y tantos recursos que causa un grave deterioro ambiental.

Posiblemente los problemas ambientales nunca se puedan controlar mientras la población humana siga creciendo a este ritmo. La predicción de incremento de nuestra población es de 2500 millones de personas para el año 2050 (UN 2006). La insistencia en un estilo de vida de alta energía magnifica el peso de la población humana sobre los recursos mundiales y sobre la calidad del ambiente.

La nueva ética debe repartir costes y beneficios entre individuos y la sociedad en su conjunto y entre las generaciones actuales y las generaciones futuras. Un mundo sostenible necesitará una ética que se incorpore dentro de la cultura. Un cambio de este tipo requerirá un notable giro en el pensamiento político y social. Más que un cambio es necesaria una revolución, que sólo se producirá si nos damos cuenta de que al dañar la Naturaleza estamos realmente perdiendo algo de valor. La clave es la educación.

Pero un mundo en el que no somos capaces de garantizar los derechos humanos más básicos, ya que se siguen incumpliendo en multitud de países 60 años después de su Declaración

Universal, no se si será capaz de garantizar un medio ambiente adecuado para las generaciones futuras. Además, no podemos olvidar que para muchas personas los problemas prácticos de la supervivencia personal hacen difícil ver la Naturaleza de una forma que no sea como fuente de alimento y combustible. Mientras, para otros la moralidad es dictada por la codicia personal más que por la preocupación por los demás, sean humanos o no.

El movimiento conservacionista moderno nació a finales del siglo XIX entre la élite social europea y norteamericana, motivados por el deseo de preservar lugares de especial relevancia estética y por la aceptación de la responsabilidad moral de los humanos para asegurar la supervivencia de las especies amenazadas (Varillas 2005). Los científicos de los años sesenta impulsaron dicho movimiento, transmitiendo lo que iban averiguando sobre la problemática situación de la fauna y flora silvestres. Algunas de sus obras causaron gran impacto a escala mundial y despertaron muchas conciencias en la década de los setenta, lo que motivó la movilización y organización de la opinión publica en asociaciones conservacionistas. Después del trabajo de los científicos, la difusión de las ideas conservacionistas debió su éxito a los medios de comunicación y en particular a destacados divulgadores como Félix Rodríguez de la Fuente, Jacques Yves Cousteau o David Attenborough. La presión conjunta de científicos y conservacionistas tuvo como consecuencia la reacción de los gobiernos, que en la década de los ochenta comenzaron a construir un tejido de instituciones y legislación relacionado con la conservación de la Naturaleza. La aplicación de la normativa y la constatación de que los ciudadanos que claman por evitar la degradación del ambiente son también consumidores, ha hecho que las empresas se incorporen también al escenario ambiental. Esta mezcla de científicos, ONGs, políticos, funcionarios y empresarios tuvo su gran momento en la Cumbre del Medio Ambiente de Rio de Janeiro en 1992. De esta reunión salieron acuerdos con importantes expectativas en los que se ha avanzado poco. Pero es indudable que en las últimas décadas se han producido algunos cambios significativos.

Esto demuestra el poder de la sociedad civil, pero también la lentitud en lograr grandes retos. Ante la situación actual, somos los ciudadanos los que debemos tomar ya la iniciativa sin esperar a que los dirigentes políticos y responsables económicos nos marquen el camino. Resignarse ante los acontecimientos o esperar a que exista otro mundo mejor en el más allá puede ser más cómodo y consolador que luchar por cambiar las cosas, pero el futuro depende de la actitud de cada uno de nosotros. El equilibrio de la vida en la Tierra depende de las plantas, insectos y bacterias, por lo que la certeza del futuro de la vida está asegurada, con los seres humanos o sin ellos. Ahora debemos tomar algunas decisiones con respecto a nuestra participación continua en ese equilibrio, participación que es claramente un privilegio y no un derecho. Es posible que ya sea demasiado tarde y que esta senda nos conduzca a un precipicio, pero de vez en cuando bajan las estrellas al camino, modificando el rumbo de nuestros sueños, y se producen cambios que parecían imposibles, cambios conseguidos tras mucho esfuerzo, cambios que cambian el mundo y que mantienen viva la esperanza. Estamos obligados a no perderla y pensar que "*yes, we can*".

Referencias

Araujo, M.B. 2004. Matching species with reserves-uncertainties from using data at different resolutions. Biological Conservation 118, 533-538.

Balmford, A., Mace, G.M. & Ginsberg, J. 1998. The challenges to conservation in a changing world: putting processes on the map. In: Mace, G.M., Balmford, A. & Ginsberg, J. (eds.) Conservation in a Changing World. Cambridge University Press, Cambridge, U.K.

Balmford A., Crane, P., Dobson, A., Green, R.E. & Mace, G.M. 2005a. The 2010 challenge: Data availability, information needs and extraterrestrial insights. Philosophical Transactions of the Royal Society B-Biological Sciences 360, 221-228.

Balmford, A., Bennun, L., ten Brink, B., Cooper, D., Cote, I.M., Crane, P., Dobson, A., Dudley, N., Dutton, I., Green, R.E., Gregory, R.D., Harrison, J., Kennedy, E.T., Kremen, C., Leader-Williams, N., Lovejoy, T.E., Mace, G., May, R., Mayaux, P., Morling, P., Phillips, J., Redford, K., Ricketts, T.H., Rodriguez, J.P., Sanjayan, M., Schei, P.J., van Jaarsveld, A.S. & Walther, B.A. 2005b. The Convention on Biological Diversity's 2010 target. Science 307, 212-213.

Brown, J.H. & Lomolino, M.V. 1998. Biogeography, 2nd edn. Sinauer Press, Sunderland, Massachusetts.

Chapin III, F.S., Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L., Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E., Mack, M.C. & Díaz, S. 2000. Consequences of changing biodiversity. Nature 405, 234-242.

Failing, L. & Gregory, R. 2003. Ten common mistakes in designing biodiversity indicators for forest policy. Journal of Environmental Management 68, 121-132.

FAO (Food and Agriculture Organization). 2008. The State of Food Insecurity in the World 2008. Available at: www.fao.org/docrep/011/i0291e/i0291e00.htm.

Fraga, J. 2006. Local perspectives in conservation politics: the case of the Ría Lagartos Biosphere Reserve, Yucatán, México. Landscape and Urban Planning 74, 285-295.

Harris, G.M., Jenkins, C.N. & Pimm, S.L. 2005. Refining biodiversity conservation priorities. Conservation Biology 19, 1957-1968.

Hawkins, B.A., Porter, E.E. & Diniz-Filho, J.A.F. 2003. Productivity and history as predictors of the latitudinal diversity gradient of terrestrial birds. Ecology 84, 1608-1623.

Hezri, A.A. & Dovers, S.R. 2006. Sustainability indicators, policy and governance: Issues for ecological economics. Ecological Economics 60, 86-99.

Hurlbert, A.H. & Jetz, W. 2007. Species richness, hotspots, and the scale dependence of range maps in ecology and conservation. Proceedings of the National Academy of Sciences 104, 13384-13389.

IPCC. 2007. Summary for Policymakers. In: Climate Change 2007: The Physical Science Basis. Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. & Miller, H.L. (eds.). Cambridge University Press, Cambridge and New York.

Koleff, P. & Gaston, K.J. 2002. The relationships between local and regional species richness and spatial turnover. Global Ecology and Biogeography 11, 363-375.

Lindenmayer, D.B., Hobbs, R.J., Montague-Drake, R., Alexandra, J., Bennett, A., Burgman, M., Cale, P., Calhoun, A., Cramer, V., Cullen, P., Driscoll, D., Fahrig, L., Fischer, J., Franklin, J., Haila, Y., Hunter, M., Gibbons, P., Lake, S., Luck, G., MacGregor, C., McIntyre, S., MacNally, R., Manning, A., Miller, J., Mooney, H., Noss, R., Possingham, H., Saunders, D., Schmiegelow, F., Scott, M., Simberloff, D., Sisk, T., Tabor, G., Walker, B., Wiens, J., Woinarski, J. & Zavaleta, E. 2008. A checklist for ecological management of landscapes for conservation. Ecology Letters 11, 78-91.

Lomolino, M.V. 2004. Conservation biogeography. Frontier of Biogeography: new directions in the geography of nature (ed. by M.V. Lomolino & L.R. Heaney), pp. 293-296. Sinauer Asoociates, Sunderland, Massachusetts.

Mace, G.M. & Baillie, J.E.M. 2007. The 2010 biodiversity indicators: Challenges for science and policy. Conservation Biology 21, 1406-1413.

Margules, C.R. & Pressey, R.L. 2000. Systematic conservation planning. Nature 405, 243-253.

Moore, J., Balmford, A., Allnutt, T. & Burgess, N. 2004. Integrating costs into conservation planning across Africa. Biological Conservation 117, 343-350.

Myers, N., Mittermeler, R.A., Mittermeler, C.G., da Fonseca, G.A.B. & Kent, J. 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853-858.

Prendergast, J.R., Quinn, R.M. & Lawton, J.H. 1999. The gaps between theory and practice in selecting nature reserves. Conservation Biology 13, 484-492.

Pressey, R.L., Cowling, R.M. & Rouget, M. 2003. Formulating conservation targets for biodiversity pattern and process in the Cape Floristic Region, South Africa. Biological Conservation 112, 99-127.

Rahbek, C. & Graves, G.R. 2000. Detection of macro-ecological patterns in South American hummingbirds is affected by spatial scale. Proceedings of the Royal Society of London, Series B 267, 2259-2265.

Rahbek, C. 2005. The role of spatial scale and the perception of large-scale species-richness patterns. Ecology Letters 8, 224-239.

Sala, O.E., Chapin III, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., LeRoy Poff, N., Sykes M.T., Walker, B.H., Walker, M. & Wall, D.H. 2000. Global biodiversity scenarios for the year 2100. Science 287, 1770-1774.

Tilman, D. 2000. Causes, consequences and ethics of biodiversity. Nature 405, 208-211.

UN. 2006. World Population Prospects: The 2006 revision. Available at www.un.org/esa/population/publications/wpp2006/wpp2006.htm.

UNDP. 2000. United Nations Millennium Declaration. Session 55/2, Resolution adopted by the General Assembly. Available at www.un.org/millennium.

UNEP. 2002. "Report on the Sixth Meeting of the Conference of the Parties to the Convention on Biological Diversity (UNEP/CBD/COP/6/20/Part 2) Strategic Plan Decision VI/26"; available at www.biodiv.org/doc/meetings/cop/cop-06/official/cop-06-20-part2-en.pdf.

Van Jaarsveld, A.S., Freitag, S., Chown, S.L., Muller, C., Koch, S., Hull, H., Bellamy, C., Krüger, M., Endrödy-Younga, S., Mansell, M.W. & Scholtz, C.H. 1998. Biodiversity assessment and conservation strategies. Science 279, 2106-2108.

Varillas, B. 2005. Participación pública y conflictos en la recuperación de especies. Una visión personal. En Jiménez Pérez, I. y M. Delibes de Castro (eds.) Al borde de la extinción: una visión integral de la recuperación de la fauna amenazada en España. EVREN. Valencia, España.

Whittaker, R.J., Araújo, M.B., Jepson, P., Ladle, R.J., Watson, J.E.M. & Willis, K.J. 2005. Conservation Biogeography: assessment and prospect. Diversity and Distribution 11, 3-23.

Capítulo 7

La utopía está en el horizonte. Me acerco dos pasos, ella se aleja dos pasos. Camino diez pasos y el horizonte se corre diez pasos más allá. Por mucho que yo camine, nunca la alcanzaré. ¿Para que sirve la utopía? Para eso sirve: para caminar.

Eduardo Galeano

Capítulo 7

Conclusiones

1. El Índice Combinado de Biodiversidad desarrollado en este trabajo fue, junto a la rareza de especies, el criterio que mejor representó la diversidad de vertebrados terrestres. El Índice Combinado de Biodiversidad destaca, además, por su alto valor intrínseco y su mayor efectividad para incluir a las especies amenazadas. A pesar de que la riqueza de especies es el criterio usado con mayor frecuencia nuestros resultados indicaron que es relativamente poco efectivo.

2. La congruencia entre las áreas relevantes de diversidad identificadas según los distintos criterios y grupos taxonómicos de vertebrados fue baja o moderada en casi todos los casos. Este hecho dificulta el desarrollo de estrategias de conservación. Aquellas estrategias basadas en un sólo criterio o que tengan en cuenta un sólo grupo taxonómico pueden proveer inadecuada protección para muchos organismos.

3. Se identificaron tres grupos de factores ecológicos y evolutivos que influyen en la distribución de las áreas relevantes de diversidad en España peninsular y Baleares: 1) los efectos biogeográficos provocados por las diferencias climáticas entre las regiones de clima atlántico y mediterráneo, y el efecto insular de las Baleares, 2) el efecto refugio de las zonas de montaña y 3) las diferencias en los requerimientos ecológicos de los distintos grupos taxonómicos.

4. Existe un número notable de áreas relevantes de diversidad que no están incluidas en la Red de Espacios Naturales Protegidos a nivel nacional y regional. Las áreas propuestas para formar parte de la Red Natura 2000 en Castilla-La Mancha tampoco incluyeron todas las áreas relevantes de diversidad, a pesar de la gran extensión de esta red.

5. La coincidencia entre las infraestructuras viarias y embalses planeados en la Península Ibérica y las áreas relevantes de diversidad de herpetofauna fue baja. No obstante, las zonas de distribución de varias especies de anfibios y reptiles serían afectadas considerablemente, con el consiguiente riesgo de pérdida de poblaciones. Consideramos necesaria alguna forma de evaluación estratégica que asegure que el desarrollo de infraestructuras sea compatible con la conservación de los hábitats y las especies.

6. Las áreas de conservación de Castilla-La Mancha no representan adecuadamente los hábitats formados por los cultivos tradicionales, a pesar de que en la región estos hábitats confieren una alta heterogeneidad al paisaje y son importantes para la conservación de la biodiversidad, especialmente para algunas especies de aves amenazadas en el contexto europeo.

7. Nuestra metodología para identificar áreas relevantes de diversidad es de utilidad para la planificación sistemática de la conservación. La combinación de dichas áreas relevantes, las áreas de conservación existentes y nuevas áreas de conectividad permite maximizar la representación de la biodiversidad de una forma eficiente, de manera que la configuración espacial resultante incluya a todas las especies y hábitats considerados importantes para su conservación.

8. Las zonas de monte mediterráneo resalveadas incrementan su diversidad estructural, proporcionando nichos adecuados para que sean ocupados por especies de aves ausentes en las zonas de monte denso. Aunque algunas de las especies de aves características del monte denso disminuyen su densidad, ninguna de ellas desapareció. Ello resulta en un aumento de la riqueza local de especies de aves que apoya la hipótesis de la perturbación intermedia.

9. El resalveo afectó a la composición de los gremios de aves, aumentando la densidad de los granívoros y disminuyendo la densidad de los insectívoros, los frugívoros y la de las especies que se alimentan en el follaje. También fue beneficioso para la persistencia en la región de las poblaciones de aves amenazadas en el contexto europeo y para las especies cinegéticas, lo que aumenta el valor de conservación y económico de este tratamiento silvícola. Recomendamos mantener zonas de monte denso para favorecer las poblaciones de sus especies características.

10. La manipulación experimental a gran escala de la estructura del hábitat y el volumen de la vegetación nos ha permitido demostrar, el predicho efecto alométrico de la complejidad estructural del hábitat en la masa corporal media de una comunidad de aves. Esto puede explicarse si asumimos que la complejidad estructural y la densidad de la vegetación actúa como un filtro selectivo de la avifauna regional, favoreciendo el asentamiento y aumento de la densidad de especies de pequeño tamaño en hábitats densos y complejos debido a limitaciones de maniobrabilidad.

11. En esta Tesis Doctoral se muestra un conjunto de herramientas e información de interés para la conservación de la Naturaleza. Más allá de su valor científico, el éxito de esta Tesis residirá en conseguir que parte de ella pueda tener alguna aplicación en el mundo real. Ello dependerá, en parte, de nuestra capacidad para transmitir eficazmente los avances de conocimiento a la sociedad en general y a los gestores y tomadores de decisiones en particular.