EFICACIA DE MEDIDAS COMPENSATORIAS PARA LA CONSERVACIÓN DE ESTEPAS AGRÍCOLAS EN ÁREAS IMPORTANTES PARA LAS AVES DEL CENTRO PENINSULAR

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TESIS DOCTORAL

2015

A mi familia y amigos

A Arantza

UNIVERSIDAD AUTÓNOMA DE MADRID

FACULTAD DE CIENCIAS

DEPARTAMENTO DE ECOLOGÍA

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Memoria presentada por Carlos Ponce Cabas para optar al grado de Doctor en Ciencias Biológicas

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Madrid, 2015

"Hemos reseñado el daño que causa la connotación negativa de los espacios esteparios españoles de singular belleza e interés científico..., importantes recursos naturales que merecen urgente protección y mejor conocimiento de quienes, mirando sin ver, los menosprecian."

Fernando González Bernáldez

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Índice

AGRADECIMIENTOS

He intentado que en este apartado aparezcan todas las personas que han hecho posible que esta tesis llegue a buen puerto, aunque es probable que me deje a algunos. Espero que sean pocos y no me lo tengan en cuenta. Simplemente no puedo acordarme de todas las personas que han participado, con su apoyo, conocimientos o en cualquier otro aspecto.

Recreación de conversación mantenida con mi primo Jandro, el gallego, hace un eón de años. Probablemente no fuera así exactamente, pero a mí me gusta recordarla de esta manera.

Jandro: *primo, vamos a ir a ver pájaros en una laguna de mi instituto. ¿Te vienes?*

Yo: *¿Ver pájaros? ¿Qué es eso?*

Jandro: *pues vamos con unos prismáticos y buscamos pájaros para saber de qué especie son.*

Yo: *pues es bastante raro lo que dices, pero os acompaño que si no tu padre me pondrá a cortar zarzas del camino.*

Un rato después...

Jandro: *¡mirad, allí hay una polla de agua!*

Yo: *jajajaja. ¿Una polla? Jajajajaja.*

Jandro: *fíjate, tiene el pico rojo y parece una gallina. Míralo en la guía...*

Desde aquel mismo momento en que miré la foto y vi el "pájaro" ese del que hablaba mi primo supe a qué quería dedicar el resto de mi vida. Me pareció maravilloso -y de locos- saber que "no todos los pájaros son iguales" y que alguien se había dedicado a sacar fotos de estos animalillos para publicarlas en un libro. Así que primo jandro, tienes que saber que mi primer agradecimiento es para ti. Gracias a ese momento en la lagunilla de tu instituto soy ahora lo que soy, un "bichólogo". Tú eres Doctor en biología y, aunque no somos de la misma rama, sabes lo que cuesta desarrollar una tesis.

Han pasado muchos años desde entonces y muchas cosas han cambiado en mi vida, pero acabar esta tesis es, probablemente, lo mejor que me ha pasado a nivel profesional.

Es obligatorio agradecer el enorme trabajo realizado por mis directores de tesis, Juan Carlos Alonso y Luis Miguel Bautista. Os agradezco a ambos la paciencia que habéis tenido conmigo en algunos momentos. Desde luego, lo que sé sobre ciencia es, básicamente, gracias a vosotros dos. Juan Carlos me permitió incorporarme al Museo Nacional de Ciencias Naturales con una beca para hacer trabajo de técnico. En ese momento no me creía que fuera a trabajar en tal institución, y encima con avutardas. Solicitamos varias becas externas al equipo para hacer la tesis, pero me rechazaban por no tener un expediente "adecuado". Esta tesis es la muestra de que se equivocaron. Aunque la beca, y posterior contrato, que me proporcionó Juan Carlos, estaba ligada a trabajo de técnico, me permitió compaginarlo con la tesis, no sin grandes esfuerzos. Ambos han hecho posible el comienzo y la finalización de esta tesis. A pesar de todo el trabajo de mis directores quiero aclarar que cualquier error que se encuentre en esta memoria de tesis doctoral es únicamente achacable a mí.

Mario Díaz me asesoró a la hora de decidir qué tipo de muestreo era más conveniente hacer en el campo para las semillas y artrópodos, una de las bases fundamentales de esta tesis. Me informó de un curso sobre medidas agroambientales que se iba a hacer. Ese curso me fue de gran utilidad para desarrollar diferentes aspectos de la tesis. Espero que haya sabido plasmarlos de una forma adecuada.

Quiero agradecer la labor de Javier Seoane, por acceder a ser mi tutor en la UAM y por su ayuda con algunos de los análisis llevados a cabo en esta tesis. Nos conocemos desde hace muchos años y es un referente para mí en el tema de las aves y la investigación.

Desde luego, esta tesis nunca habría podido llegar a buen término sin la participación de los agricultores en el Programa de Medidas Agroambientales. A pesar de algunos momentos complicados, siempre habéis hecho lo posible para que nuestros experimentos pudieran llevarse a cabo.

Quiero agradecer también a los primeros avutarderos con los que trabajé en el museo. A Carlos Martín, Carlos Palacín, Marina Magaña, Beatriz Martín y Pablo Sastre por los ratos que pasamos juntos durante años, con las avutardas y con otras especies. Algunos de vosotros habéis colaborado en gran medida en varios capítulos de la tesis. Carlos Martín y Carlos Palacín hicieron posible la señalización de los tendidos eléctricos y Beatriz los recorrió en búsqueda de cadáveres. Marina fue la responsable del Programa de Medidas Agroambientales hasta que yo asumí esa labor. Con Pablo compartí bastantes jornadas de campo registrando molestias a las avutardas.

También me gustaría expresar mi gratitud hacia mis últimos compañeros en el museo. Iván salgado y Aurora Torres, con los que he vivido inolvidables momentos en el campo. Iván ha sido mi compañero de andanzas en el capítulo "de los huevos" y Aurora siempre me ha ayudado con el R y el GIS, desde que un revisor me pidió que hiciera un modelo mixto -me sonaba a un sandwich- en lugar de una prueba de datos pareados, hasta enviarme comandos de R. Aurora y yo hemos compartido muchas horas de censo y de capturas, mucho más agradables que si fuera yo solo. Estoy seguro de os espera a ambos un futuro muy bueno en el mundo científico.

Varios estudiantes han participado de forma notable durante diferentes fases de esta tesis. Quiero agradecer especialmente su colaboración a Elena, Gonzalo, Natalia, Desi, Dácil, Iris y Alberto.

Un agradecimiento especial debe ir a Carolina Bravo, con quien he compartido muchas horas caminando por el campo con una cinta métrica atada a la espalda o hablando de "las lolas". Recuerdo sobre cualquier otra cosa las quedadas con el todoterreno y los calores que pasamos en algunos lugares, como en Aranjuez, que desembocaron en problemillas leves de salud. Gracias a tu aportación se han podido llevar a cabo casi todos los capítulos de esta tesis (y algunos artículos que vendrán después). Espero poder seguir colaborando contigo en el futuro.

No puedo olvidarme de Rafael Barrientos y de Jose Manuel Álvarez. Cada uno habéis estado ahí en diferentes momentos. Con Rafa he compartido trabajo de

campo y gabinete durante años, mucho más tiempo de lo que duró su corta vinculación al equipo. Jose me ayudó en algunos momentos de la tesis, con análisis variados. He aprendido mucho de ti. Desde luego, con ambos he pasado también momentos "distendidos" dentro y fuera del museo.

Antes de entrar a trabajar en el Museo Nacional de Ciencias Naturales pude formarme como biólogo y como ornitólogo. Debo agradecer a un profesor en particular la forma de enseñar y su accesibilidad hacia los alumnos. Como no podía ser de otra forma, me refiero a Quico. La asignatura que él impartía era una maravilla. Cuando le preguntábamos, él parecía encantado, así que nosotros seguíamos preguntando, con ciertas repercusiones sobre el temario, pero mereció la pena. Las salidas de los cursos de doctorado eran "especiales"., con grandes variaciones sobre lo previsto. Gracias Quico por ofrecerme en su momento un contrato para trabajar con las alondras de dupont (ricotí no me gusta). Tú me animaste a pedir la beca que había salido en el museo para trabajar con las avutardas. Así lo hice, y te lo agradeceré siempre. Recuerdo también cuando te dije mi apellido. En ese momento te cambió la cara. Eso de saber que tú y mi padre trabajasteis juntos en tus comienzos me hizo mucha ilusión. Quico, tu legado en forma de publicaciones y actuaciones diversas para conservar este grupo de "pájaros marrones" al que llamamos esteparias es encomiable. Ningún amante de las aves podrá olvidarte.

Mi formación como pajarero empezó (y continua actualmente) de la mano de Monticola. Con ellos he aprendido lo que sé sobre aves. Quiero agradecer especialmente a Juancho sus horas "no muertas" hablando de aves, y anillando para sacar fichas de muda que se tradujeron en diversos trabajos. ¿Cuántas aves hemos anillado tú y yo juntos? Es una lástima que ahora estés tan lejos. También es necesario dar las gracias a otros monticoleños (o monticolocos) con los que también he disfrutado mucho. Cristian, Óscar, Virginia, con vosotros he compartido innumerables horas de campo, de día y de noche. Me alegro de que todos forméis parte de mi vida.

Mis años en el museo han sido muy especiales por toda la gente que he conocido. Cada uno dedicado a un tema diferente, pero entre todos hacemos un grupo "curioso". Quiero agradeceros a todos vuestra amistad, que espero dure

hasta que las ranas tengan melena. Seguro que me olvido de algunos, pero muchas gracias a Rigo, Jose, Chechu, Diego, Elisa, Chio, Dani, Marcos, Marti, Sergio, Jimena, Reimon, Jaime, Eva, Roger, Shirin, Geizi, Natalia y el resto de pestuz@s del museo. ¡¡Que siga la fiesta!!!

Gracias a mis compañeros de la 1111, con los que he tenido muchos momentos inolvidables dentro del despacho, incluida alguna que otra fiestecilla. Antón, Ibáñez, Isaac, Juan, Octavio, Rafa, Cantarero (como si fueras de la 1111) y, por qué no decirlo, Regan, entre todos habéis hecho que mi estancia en ese despacho haya sido muy agradable.

No quiero olvidarme de mi familia. Mis hermanos y mis padres, entre todos me habéis animado a hacer lo que siempre quise, trabajar con aves.

Por último, quiero agradecer el apoyo a la persona más importante de mi vida, Arantza. Me has apoyado y animado en momentos muy duros y hemos compartido nuestra pasión por las aves. Me has acompañado al campo, a echar horas muestreando, censando y con los documentos finales de esta tesis. Siempre has estado ahí, conmigo. ¡¡GRACIAS!!

INTRODUCCIÓN GENERAL

Las estepas son áreas abiertas con vegetación escasa y generalmente de porte bajo (Valverde 1958). Sin embargo, con frecuencia también se denominan pseudoestepas o estepas agrícolas o cerealistas al complejo de hábitats agrarios de cereal de secano de gran extensión. La variación en el nombre se produce para diferenciarlas de las verdaderas estepas del este de Europa, que no tienen aprovechamiento agrícola. En todos los casos estos espacios poseen las siguientes características:

- Simplicidad estructural: Tienen una cubierta vegetal que incluye solamente herbáceas o arbustos de pequeño porte.

- Buena visibilidad, como consecuencia de la escasez de vegetación de gran porte

- Ausencia de lugares de nidificación protegidos (como árboles o acantilados)

- Necesaria exposición a las inclemencias climáticas, debido a la escasez de refugio: viento, lluvia, insolación

- Altas fluctuaciones en las temperaturas entre diferentes estaciones, y especialmente entre el día y la noche

- Escasez de zonas provistas de agua de manera permanente

- Gran dinamismo, puesto que se producen grandes modificaciones del paisaje en poco tiempo

En la Península Ibérica podemos distinguir tres grandes unidades (Suárez *et al.* 1991): las **estepas leñosas**, dominadas por arbustos de pequeño porte, fundamentalmente caméfitos, entre los que suceden frecuentes calveros de suelo desnudo. En esta categoría también se incluyen las comunidades halófilas con elementos subarbustivos, espartales y albardinales. Por otro lado, se encuentran los **pastizales**, llanuras densamente cubiertas de comunidades herbáceas entre las que se intercalan zonas de matorral más alto y pies aislados de arbolado. Finalmente, la **estepa cerealista**, que se compone de territorios llanos o ligeramente ondulados dedicados en su mayoría al cultivo de cereal en secano.

Con la denominación de *aves esteparias* se conoce a un grupo de aves que tienen en común el desarrollo de la totalidad o parte de su ciclo biológico en un hábitat determinado, el medio estepario. Estas aves son un grupo heterogéneo desde un punto de vista filogenético y sistemático que, sin embargo, presentan similares adaptaciones morfológicas, fisiológicas, etológicas y ecológicas que favorecen su supervivencia en este medio y que resultan ejemplos de convergencia adaptativa. La presencia de aves esteparias en nuestra región data de antiguo, y está documentada en el registro fósil (Santos & Suárez 2005).

Estas especies de aves han estado capacitadas para su expansión, independientemente de su origen, gracias a una mayor dependencia de la estructura del paisaje frente a caracteres edáficos o florísticos, lo que justifica la colonización de medios tan humanizados, especialmente los sujetos a regímenes de explotación tradicionales y cierta diversidad paisajística. El cultivo de cereal de secano ofrece mejores condiciones de termorregulación para la nidificación de algunas aves esteparias que las ofrecidas por la vegetación natural de las estepas (Farago 1986).

Para avalar la importancia que tienen estos paisajes agrícolas para la avifauna ibérica cabe decir que el 27% de las IBA —acrónimo inglés de *Áreas Importantes para las Aves*— de nuestro territorio presentan medios típicamente agrícolas y que la Península Ibérica tiene la mejor representación de aves esteparias de la Unión Europea, donde los agrosistemas ocupan casi la mitad de su superficie (European Comission, 2011). En la Península Ibérica destaca la importancia de cultivos cerealistas de secano, presentes en un 22% de las IBA, que resultan esenciales para el desarrollo de gran parte del ciclo biológico del muchas especies de aves. Dichos sistemas agrícolas están muy ligados al ser humano y por ello están expuestos a ciclos muy dinámicos que implican alteraciones muy acentuadas sobre el entorno en períodos de tiempo muy cortos como la recogida del cereal, laboreo de las tierras, etc. En definitiva, no es extraño que la adecuada conservación de los ecosistemas peninsulares pase por un apropiado mantenimiento de los sistemas agrarios tradicionales, donde el ser humano no sólo no es dañino, sino que es fundamental para su conservación.

Actualmente, los taxones ligados a estos medios se enfrentan a problemas de diversa índole que se pueden agrupar en la destrucción, alteración y fragmentación de su hábitat. Es necesario recordar que estas amenazas genéricas son la principal causa de pérdida de biodiversidad a nivel mundial no sólo en las estepas de otros continentes, sino en la mayoría de los ecosistemas del planeta. La multitud de problemas de las estepas superan las posibilidades temporales y materiales de una tesis doctoral para ser tratados adecuadamente. En la presente tesis nos centraremos en dos aspectos relacionados con los impactos en las aves esteparias: la mortalidad de aves en tendidos eléctricos y la intensificación agrícola.

Problemática asociada a los tendidos eléctricos

El transporte de electricidad desde las plantas a los usuarios se produce habitualmente vía aérea mediante líneas eléctricas. Sin embargo, éstas provocan la muerte de gran cantidad de aves, tanto por electrocución como por colisión (además de otros problemas de diversa índole (Ferrer 2012). En los Estados Unidos de América se estima que mueren más de 175 millones de aves como consecuencia de las colisiones y electrocuciones en tendidos eléctricos (Manville 2009).

En España, aunque se desconoce la magnitud real del problema a escala nacional, sí se sabe que las muertes por colisión contra tendidos eléctricos suponen un grave problema, para las aves en general y para las aves esteparias en particular (Alonso *et al*. 1994, Palacín *et al*. 2004) y, en 2008 se elaboró un Real Decreto (Real Decreto 1432/2008), por el que se establecen medidas para la protección de la avifauna contra la colisión y la electrocución en líneas eléctricas de alta tensión.

Además, las colisiones contra tendidos eléctricos son la principal causa conocida de mortalidad no natural en la Avutarda común (*Otis tarda*, Palacín *et al*. 2004), la especie más representativa de las estepas. Dichas muertes tienen un grave impacto en la dinámica poblacional de la especie, hasta el punto de que es el principal factor influyente en la probabilidad de extinción de las avutardas en varios lugares del centro peninsular (Martín 2008). Sin embargo, el impacto de los

tendidos eléctricos no se traduce exclusivamente en la muerte directa de las aves colisionadas o electrocutadas, sino que también pueden cambiar patrones comportamentales (Sergio *et al.* 2004).

Desde hace años, las investigaciones sobre la viabilidad poblacional de las aves afectadas han tratado de mitigar las muertes de aves mediante medidas correctoras. Para el caso de la colisión, la medida más habitual es la incorporación de dispositivos anticolisión en los cables, ya sea el cable de tierra o en los propios conductores (Ferrer 2012). El hecho de colocar los dispositivos en el cable de tierra se debe a que se le considera responsable de la mayor parte de las colisiones debido a su menor diámetro (Heijnis 1980, Beaulaurier 1981). Estos dispositivos aumentan la visibilidad de los cables, de manera que las aves puedan evitar la colisión.

Sin embargo, existen pocos estudios diseñados correctamente para obtener conclusiones sobre la eficacia de los dispositivos. Los estudios suelen carecer de un tamaño muestral suficiente o de un correcto diseño experimental (Barrientos *et al.* 2011). Además, la mortalidad real producida se subestima al no incorporar factores externos a las propias colisiones que, sin embargo, influyen en la estima de su frecuencia (Bevanger 1999). Dichos factores son:

1.- la detectabilidad debido a la experiencia de los observadores y características del hábitat

2.- imposibilidad de muestrear en determinados lugares

3.- la desaparición de los cadáveres colisionados por la acción de animales carroñeros

Problemática asociada a la intensificación agrícola

La agricultura ha experimentado tras la Segunda Guerra Mundial un notable proceso de intensificación en la mayor parte de los países Europeos (Gardner 1996, Robinson & Sutherland 2002). Dicha intensificación se define como el aumento de la producción agrícola por unidad de superficie (Donald *et al.* 2001).

Para lograr ese incremento en la producción, la agricultura ha cambiado notablemente (Sans *et al.* 2013). Así, nos encontramos en la actualidad con:

incremento en los insumos utilizados (fertilizantes sintéticos, biocidas), pérdida de heterogeneidad de cultivos (tanto en especies como en variedades empleadas), puesta en cultivo de terrenos incultos, concentración parcelaria (con la consiguiente eliminación de bordes entre parcelas), un aumento en los trabajos de laboreo en las zonas agrícolas (que dificultan la presencia de plantas silvestres, invertebrados y fauna vertebrada), el laboreo de parcelas recién cosechadas, descenso en la anchura de los bordes entre parcelas, siembra de variedades de ciclo corto, y tendencia a cosechar en épocas más tempranas.

Las consecuencias directas de la intensificación agrícola han sido ciertos costes en términos ecológicos. Fundamentalmente, se ha producido una gran reducción de la biodiversidad en el medio agrícola de diferentes grupos taxonómicos (Clough *et al.* 2005, Fuller *et al.* 2005, Donald *et al.* 2006), incluyendo aves, mamíferos, artrópodos y plantas silvestres (ver revisión en Benton *et al.* 2003).

En particular, se han detectado disminuciones alarmantes en muchas especies de aves que, gracias al seguimiento que hacen los países de sus poblaciones, son buenas indicadoras de las variaciones en calidad del hábitat.

Según los resultados obtenidos por el European Bird Census Council (EBCC 2010), las aves ligadas a ambientes agrarios están en claro retroceso a nivel Europeo. De las 36 especies de aves consideradas para el último cálculo de su tendencia (EBCC 2014), 26 muestran una tendencia negativa. De ellas, 19 tienen una tendencia de "declive moderado" (menor del 5% anual), y 3 un declive fuerte (mayor del 5% anual). Según el EBCC, existe un declive de casi el 50% para el conjunto de especies ligadas a medios agrarios.

Las investigaciones llevadas a cabo en varios países concuerdan con la información aportada por EBCC. Así, por ejemplo, en Reino Unido se detectó que en 20 años (1970-1990) el 86% de las especies de aves (de 28 estudiadas) habían reducido sus rangos de distribución, o que el 83% de 18 eran menos abundantes (Fuller *et al.* 1995). No solo eso, sino que un estudio comparativo reflejó que los tamaños poblacionales de aves eran, de media, el 52% entre los años 1968 y 1995 en algunas especies en declive (Siriwardena *et al.* 1998).

La situación en España no es más alentadora. De hecho, según Escandell *et al.* (2011), algunas de las especies están decayendo incluso a un ritmo mayor que en el conjunto de Europa. Así, nos encontramos con descensos acumulados del 26% (hasta un 72% en el norte de España) en la Calandria común (*Melanocorypha calandra*), del 20% en la Alondra común (*Alauda arvensis*) o del 72% en la Ganga ortega (*Pterocles orientalis*) desde el año 1998, momento en el que comenzó el Programa de Seguimiento de Aves Comunes Reproductoras (SACRE) de SEO/BirdLife.

Los motivos relacionados con la intensificación agrícola por los que la tendencia de las aves ligadas a estos medios es tan negativa son el descenso en la cantidad y calidad de alimento y la pérdida de lugares seguros para nidificar, con aumento en la depredación de nidos (Newton 2004).

En respuesta a estos hechos, a lo largo de las últimas décadas se han generalizado en varios países europeos diversos sistemas de primas a los agricultores, a cambio de que estos modifiquen sus prácticas agrícolas y sigan unas directrices tendentes a mejorar la conservación de la biodiversidad en el medio agrícola (Programas de Medidas Agroambientales). Así, la Política Agraria Común (PAC) de la Unión Europea introdujo en su reforma de 1985 la posibilidad de que los Estados miembros invirtiesen fondos en la conservación de zonas ambientalmente sensibles y, en 1992, se aprobó la regulación CEE 2078/92, demandando la aplicación de medidas agroambientales. Por último, en las últimas reformas de la PAC no se condiciona la cuantía de las compensaciones a la producción agrícola, sino a la superficie. Además, se condicionan las ayudas recibidas a unas normas y prácticas básicas (condicionalidad).

Sin embargo, tras un largo periodo de funcionamiento de estos programas de medidas agroambientales, no existe consenso sobre el grado de cumplimiento de su objetivo, el cual no es otro que contribuir a conservar la biodiversidad y revertir los problemas generados por la intensificación agrícola. La ausencia de conclusiones claras sobre la efectividad de los programas de medidas agroambientales deriva del hecho de que muchas de las prácticas habituales en conservación de la naturaleza se basan más en asunciones y conclusiones extraídas sin fundamento o heredadas de anteriores experiencias que en un análisis

científico de la realidad. En la revisión de Kleijn y Sutherland (2003) sólo encontraron 62 estudios de tan solo seis países en los que se examinara la eficacia de los planes de medidas agroambientales. La mayoría de esos estudios habían sido realizados en sólo dos países (Reino Unido y Holanda), sólo uno en Portugal, y ninguno en Francia ni en España, circunstancia que llamó la atención de los autores, quienes advierten reiteradas veces en su trabajo sobre la urgente necesidad de estudios de este tipo en países del entorno mediterráneo.

Por último, en muchos de los estudios no se detectó beneficio alguno para la biodiversidad tras la aplicación de las medidas agroambientales (Kleijn *et al*. 2006). Es más, en algunos (ver revisión en Kleijn & Sutherland 2003) se observaron efectos negativos sobre algunos de los grupos analizados. Así pues, hoy por hoy es cuestionable la eficacia de muchos de los programas de ayudas agroambientales vigentes, e incluso, la conveniencia de que dichos programas sigan implementándose en el futuro. Teniendo en cuenta que a lo largo de la pasada década se han invertido sólo en Europa 24 millones de euros en programas de ayuda agroambientales, es urgente la realización de un mayor número de estudios que evalúen de forma científicamente contrastada la eficacia de dichas medidas, para poder optimizar los recursos económicos que actualmente se destinan a hacer compatible la agricultura y la conservación de la biodiversidad en el medio agrícola.

En esta tesis se evalúa la eficacia del siguiente plan de medidas compensatorias:

Proyecto de Medidas preventivas, correctoras y compensatorias de la afección de la M-50 y de la Autopista de peaje R-2 a la población de avutardas y otras aves esteparias de la IBA Talamanca-Camarma, y al LIC Cuenca de los ríos Jarama y Henares

En abril de 2001, la empresa Autopista del Henares, S.A. (HENARSA), constructora de la R-2 Madrid-Guadalajara y del tramo de la M-50 con el que enlaza, encargó a un equipo del CSIC dirigido por el Profesor de Investigación J.C. Alonso el *Proyecto de medidas preventivas, correctoras y compensatorias de la afección de la M-50 (tramo M-607/N-IV, subtramo N-I/N-II) y de la Autopista de*

peaje R-2 a la población de avutardas y otras aves esteparias de la IBA Talamanca-Camarma, y al LIC Cuenca de los ríos Jarama y Henares.

La autopista de peaje R-2 atraviesa la zona sur de la ZEPA 139 "Estepas cerealistas de los ríos Jarama y Henares" en la Comunidad de Madrid y el sureste de la ZEPA 167 "Estepas cerealistas de la campiña" en la provincia de Guadalajara, mientras que la M-50 se encuentra próxima a la ZEPA 139 y enlaza con la R-2 (Figura 1).
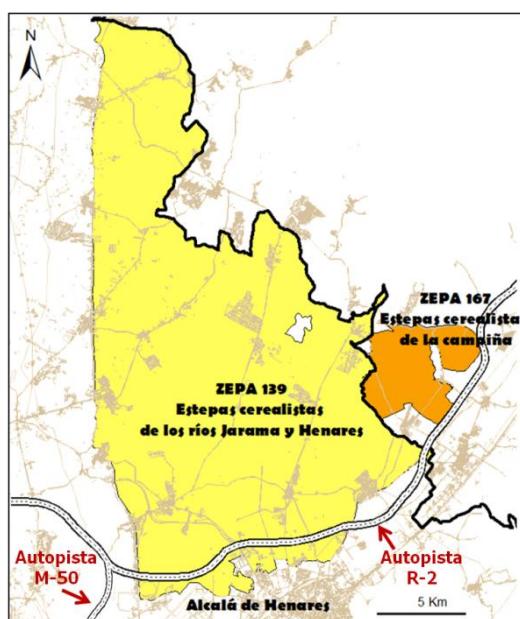
Figura 1. Zonas de especial Protección para las Aves (ZEPA) 139 y 167 y trazado de las autopista R-2 y M-50. Elaborado por Aurora Torres y modificado parcialmente en esta tesis.

En el marco de dicho Proyecto se aplicó un plan de medidas compensatorias de conservación para paliar el efecto de la construcción de ambas carreteras y compensar a las zonas agrícolas afectadas por la pérdida directa de hábitat. Dicho plan de medidas compensatorias comprende diferentes actuaciones que se pueden agrupar en:

1.- Plan de divulgación y educación ambiental en diferentes centros educativos (no tratado en la presente tesis).

2.- Actuaciones para reducir las molestias a la fauna y la mortalidad. Se llevó a cabo la señalización de tendidos eléctricos en la Comunidad Autónoma de

Madrid y en la provincia de Guadalajara. Esta actuación se desarrolla en los capítulos 1 y 2 de la presente tesis.

3.- Actuaciones para mejorar la calidad del hábitat. Se implementó un Plan de Medidas Agroambientales en la IBA Talamanca-Camarma en las provincias de Madrid y Guadalajara. Esta actuación se desarrolla en los capítulos 3, 4 y 5 de la presente tesis. Las medidas agroambientales implementadas en el Programa fueron:

   a) Mejora y mantenimiento del barbecho tradicional (mantenimiento de rastrojeras)

   b) Barbecho semillado con leguminosas

   c) Retirada de la producción de tierras

   d) Rotación de cultivos trigo- girasol

   e) Cultivo de cereal no tratado

   Las prescripciones concretas de cada medida se muestran en el Anexo 1 de la tesis.

4.- Plan de seguimiento y valoración de las medidas. Comprende toda la tesis en su conjunto.

Desarrollo de las actuaciones

El procedimiento efectuado para la señalización de los tendidos, consistió en contactar con las empresas propietarias de cada uno de los tendidos en los que se habían detectado importantes mortalidades de aves. Después del análisis de viabilidad (realizado por la empresa propietaria del tendido) se produjo la petición de ofertas a empresas fabricantes de dispositivos anticolisión. Una vez dichos dispositivos fueron colocados se procedió a evaluar su eficacia.

El procedimiento para la selección de parcelas a incluir en el Programa de Medidas Agroambientales se refleja en la figura 2.
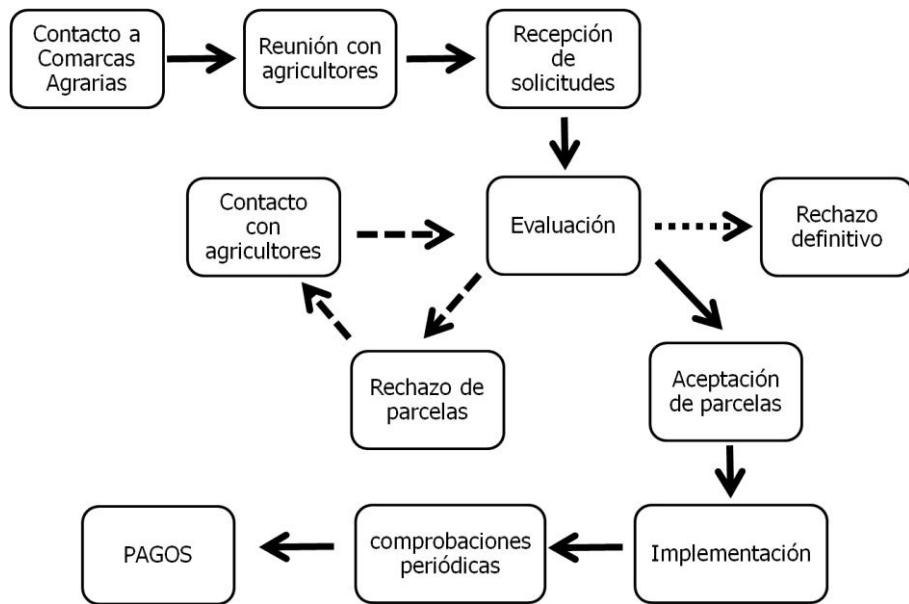
Figura 2. Procedimiento resumido del Programa de Medidas Agroambientales.

Se llevó a cabo un primer contacto con las diferentes asociaciones de agricultores en la zona de actuación. Una vez que los agricultores enviaban su solicitud de incorporación de parcelas al Programa, ésta era evaluada por el equipo dirigido por el Profesor de Investigación del CSIC Juan Carlos Alonso. En ese momento se descartaban algunas parcelas, otras se aprobaban y otras se rechazaban de forma condicional. En este último caso se volvía a contactar con los agricultores para valorar la posibilidad de cambiar el tipo de medida agroambiental a aplicar y se evaluaba la nueva propuesta. Si ésta satisfacía las necesidades del Programa se admitían. En caso contrario se rechazaban. Al implementar cada medida, se llevaban a cabo controles periódicos sobre las parcelas. Si los agricultores cumplían el compromiso adquirido se realizaba el pago de las primas correspondientes.

Objetivos y estructura de la tesis

El objetivo principal de esta tesis es evaluar la eficacia del plan de medidas compensatorias. Concretamente, nos centramos en las actuaciones 2, 3 y 4 mencionadas anteriormente.

Así pues, la tesis doctoral está estructurada en los siguientes capítulos:

Capítulo 1. Carcass removal by scavengers and search accuracy affect bird mortality estimates at power lines.

En este capítulo se desarrolla un experimento para cuantificar la desaparición de cadáveres en tendidos eléctricos debida a la acción de animales carroñeros, así como las diferencias en la detectabilidad de dichos cadáveres por diferentes observadores que difieren en su grado de experiencia de localizar aves muertas bajo los tendidos eléctricos. Ambas cuestiones son fundamentales para obtener estimas reales del impacto que tienen las líneas eléctricas sobre las aves.

Capítulo 2. Wire marking results in a small but significant reduction in avian mortality at power lines: A BACI designed study.

Mediante un diseño BACI se evalúa la eficacia de la señalización mediante espirales salvapájaros de tramos de líneas eléctricas en la Comunidad de Madrid y la provincia de Guadalajara que atraviesan zonas agrícolas de importancia para las aves esteparias. También se estudia la posibilidad de que diferentes tamaños de espiral produzcan resultados diferentes, así como la señalización de diferentes tipos de tendidos eléctricos (de transporte o de distribución).

Capítulo 3. Effects of organic farming on plant and arthropod communities: A case study in Mediterranean dryland cereal.

Se realiza un análisis de la eficacia del cultivo ecológico de cereal de secano para plantas y artrópodos. Se comparan valores de abundancia, riqueza y diversidad para ambos grupos entre diferentes formas de manejo, convencional y ecológica. También se compara la biomasa de invertebrados obtenida en ambos tipos de manejo.

Capítulo 4. Effects of agri-environmental schemes on farmland birds: do food availability measurements improve patterns obtained from simple habitat models?

En este capítulo se estudia la respuesta de las aves ante el manejo de parcelas incluidas en el Programa de Medidas Agroambientales. Concretamente, se examina la importancia de la estructura del hábitat, el paisaje y la cantidad de alimento presente en la abundancia, riqueza, diversidad y un Índice que incluye información de la lista de Especies Amenazadas de Preocupación en Europa (SPEC)

desarrollada por BirdLife International. Asimismo, se compara la utilidad y capacidad predictiva de dos tipos de modelo, uno en el que en incluyó la cantidad de alimento (modelos costosos) y otro en el que se emplearon variables de superficie (modelos sencillos).

Capítulo 5. Do agri-environmental schemes effectively protect nests from predation? An experimental study.

En este capítulo se desarrolla un experimento de depredación de nidos. Se relaciona las tasa de depredación de nidos con variables a escala de paisaje, parcela y estructura de la vegetación en torno al nido. Se emplean cámaras de fototrampeo para conocer las especies de depredador presentes y relacionar las tasas de depredación con su estrategia de búsqueda de alimento.

Después de los capítulos anteriores se presenta una discusión general de los principales resultados obtenidos y, de manera resumida, las conclusiones generales de la presente tesis doctoral.

Cada uno de los capítulos mencionados anteriormente reproduce íntegramente un manuscrito publicado o en fase de publicación. En ambos casos, su estado y la referencia completa (cuando proceda) se ha incluido al comienzo de cada uno de ellos. Las revistas poseen normas de publicación diferentes, por lo que se ha mantenido el formato de referencia, figuras o tablas de las normas editoriales de cada revista.

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CAPÍTULO 1

Este capítulo reproduce íntegramente el siguiente artículo:

Ponce, C., Alonso, J.C., Argandoña, G., García Fernández, A. & Carrasco, M. 2010. Carcass removal by scavengers and search accuracy affect bird mortality estimates at power lines. *Animal Conservation* 13(6): 603–12.

CAPÍTULO 1

Carcass removal by scavengers and search accuracy affect bird mortality estimates at power lines

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ABSTRACT

Bird mortality as a result of collisions with power lines has been of increasing concern in recent decades, but the real impact on bird populations requires an experimental assessment of scavenger removal rates and searcher detection errors. Farmland and steppe birds, two of the most threatened avian groups, have been shown to be particularly vulnerable to collision with power lines, but few removal and detectability studies have been developed in cereal farmland habitats, and none in the Mediterranean region. We conducted five carcass disappearance trials in central Spain by placing 522 corpses of different sizes under power lines, and searching for remains four times during the following month. The influence of several factors was examined using multivariate approach. The accumulated number of carcasses removed by scavengers increased logarithmically, with 32% removed over the 2-day period after the initial placement, but only 1.5% removed on a daily basis by day 28. Small birds disappeared earlier and at a higher proportion than larger birds. Carcass removal rates were site-dependent, but were not influenced by carcass density or season. The detection rate increased with the observer's previous experience and carcass size. Carcass counts at power lines notably underestimate bird casualties. Our 4-week disappearance equations provide a full range of scavenging rates and observer efficiency correction factors for a wide range of bird weights. Fortnightly to monthly search frequencies may be adequate to detect medium- to large-sized corpses, but are insufficient for smaller birds. Finally, all personnel participating in carcass searches should be trained previously in this task.

Keywords

INTRODUCTION

Electric power lines are known to be a cause of bird mortality, either through electrocution or collision with the wires (Bevanger, 1994, 1998; Ferrer & Janss, 1999; Bevanger & Broseth 2001; Erickson *et al.*, 2001; APLIC & USFWS, 2005). This has generated an increasing concern due to the negative effect it may have on some species that are particularly vulnerable to these mortality factors (Haas, 1980; Ferrer, de la Riva & Castroviejo, 1991; Alonso, Alonso & Muñoz-Pulido, 1994; Janss, 2000; Janss & Ferrer, 2000). The only efficient way to evaluate the impact of such mortality is to count dead birds in the power line corridor (Beaulaurier, 1981; Faanes, 1987; Bevanger ,1999). However, because field researchers cannot continually monitor the power lines, scavengers can be expected to find and remove a variable portion of the carcasses between the time of their deaths and the time the next search is conducted. Also, a number of the carcasses or their remains will be missed by the observers patrolling the line. Therefore, the results of carcass searches are affected by two main bias sources, (a) the rate at which carcasses are removed by scavengers, and (b) the ability of observers to detect corpses or their remains in the field.

A recent review of birds found poisoned after agricultural pesticide treatment stated that removal rates may vary widely, altering the mortality estimates based on carcass searches (Prosser, Nattras & Prosser, 2008). Among possible factors influencing removal rates are features affecting visibility of corpses such as their size, colour, or vegetation cover, and local and/or seasonal changes in scavenger abundance and activity (Heijnis, 1980; Beaulaurier, 1981; Wobeser & Wobeser, 1992; Bevanger, 1999; Morrison, 2002; Ward *et al.*, 2006). As for the searcher efficiency, it has also been shown to differ extensively with vegetation type and size of the bird (Wobeser & Wobeser, 1992; Bevanger, 1999; Morrison, 2002). Scavenger removal rates and efficiency of field workers should therefore be known to ensure that these bias sources can be corrected to obtain accurate estimates of bird mortality rates.

The objectives of this study were to (a) determine the carcass removal rate of power line collision victims and the observers' search bias by means of a series of trials simulating collisions of birds with power lines in a farmland area in central Spain, and (b) examine the influence of various potentially relevant factors such as study site and season, carcass size and density, and vegetation height and cover, using a multivariate approach. The aims were to (i) obtain correction factors for these two bias sources which may be used to improve bird fatality estimates at power lines –although correction factors obtained in our study should be applied with caution by researchers working in areas with different habitat characteristics–, and (ii) suggest acceptable periodicities to conduct future carcass searches at power lines in farmland habitat. Various studies have carried out similar carcass removal experiments (Prosser *et al.*, 2008; and references therein), but few have tried to analyse simultaneously the influence of several factors. Most of these carcass removal studies have been done to estimate mortality after pesticide treatment in North America or northern Europe (reviewed in Prosser *et al.*, 2008), some at wind turbines (reviewed in Morrison, 2002; Siriwardena *et al.*, 2007), and very few analogous studies have been published in relation with mortality at power lines (e.g. Bevanger, Bakke & Engen, 1994), although there may be unpublished reports produced by private companies that are not available. To our knowledge, this is the first study carried out specifically to assess scavenger removal rates and search efficiency of birds found dead at power lines in Mediterranean farmland habitats, using a multivariate approach to deal simultaneously with several underlying variables.

The two most commonly recognized sources of error affecting bird mortality estimates at power lines or wind turbines are carcass removal by scavengers and observer detection error (e.g. Bevanger, 1999; Siriwardena *et al.*, 2007). A widely used estimator of adjusted bird mortality (M_A) is therefore $M_A = M_U/R \times p$ where M_U is the unadjusted mortality expressed as number of fatalities per km of power line, or wind turbine per year, R is the proportion of carcasses remaining since the last fatality search, and p is the proportion of carcasses found by observers searching for dead birds. In the present study we provide a full range of correction factor values for R and p through a month after fatality occurrence, by conducting four-week long carcass disappearance trials and developing carcass

removal and searcher efficiency equations for four different carcass sizes. From these equations, correction factors for these two main bias sources can be calculated for any search periodicity up to one month between consecutive search surveys, and covering a wide range of bird weights (ca. 50-1 000 g). Other minor adjustments referring to birds injured by the wires but that die elsewhere remaining undetected (crippling bias), and natural mortality not caused by collision with the wires (background mortality) are not quantified because they are usually assumed to be relatively small.

Farmland areas host many endangered bird species which have suffered alarming population decreases during the last few decades, due mostly to agricultural intensification but also due to other human-induced causes (Tucker & Heath, 1994; Siriwardena *et al.*, 1998; Donald, Green & Heath, 2001; Wretenberg *et al.*, 2006). Among these, the ever-increasing number of power lines built on farmland areas, where terrain conditions are more suitable for the installation of these utility structures, is currently an issue of great concern. Farmland and steppe species are indeed at present the most threatened bird group, with 83% of the species subject to unfavourable status (BirdLife International, 2004; Burfield, 2005; Santos & Suárez, 2005). Many of these steppe-birds have significant yet endangered or declining populations in the Iberian Peninsula (Madroño, González & Atienza, 2004; Santos & Suárez, 2005), and some of them are particularly vulnerable to the negative effects of power lines (e.g., common cranes or great bustards, for which collision with power lines has been identified as the main cause of adult mortality, Alonso & Alonso, 1999; Janns & Ferrer, 2000; Palacín *et al.*, 2004).

METHODS

Study area

The study was conducted in five Important Bird Areas in Madrid province, along with a small area in Guadalajara province, central Spain. In each of these areas we selected 1-2 km long sectors of power lines covering 14 km of 11 different power lines in total (Figure 1). The terrain is flat to slightly undulated, with an elevation

of 740 $\pm$ 83 m.a.s.l. It is primarily dedicated to cereal cultivation (mainly wheat *Triticum aestivum* and barley *Hordeum spp.*), with minor fields of legumes (*Vicia spp. and Medicago sativa*), olive groves *Olea europaea* and grapevines *Vitis vinifera*. Most cereal is grown in a traditional two-year rotation system that creates a dynamic mosaic of ploughed, cereal and stubble patches over the region. The climax vegetation of evergreen oak trees (*Quercus ilex*) and *Retama sp.* and *Thymus sp.* Scrubland has been generally cleared up to small open-wooded tree plots interspersed within the dominant farmland. White poplars (*Populus alba*) are also found in the IBA, although as in the case of oaks, always as single trees or small groups.
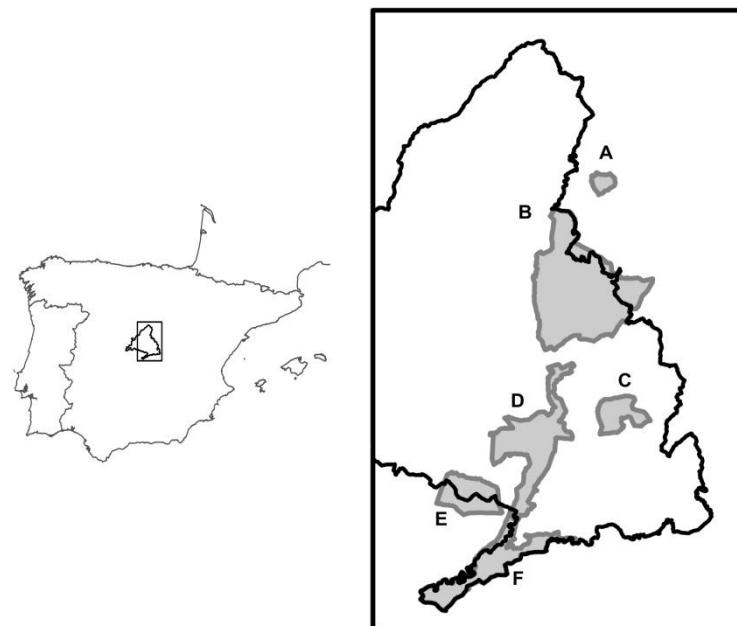
Figure 1. Location of the study area in Madrid region and number of power lines surveyed (in brackets). A: Casa de Uceda (1), B: IBA 074 Talamanca-Camarma (5), C: IBA 075 Alcarria de Alcalá (1), D: IBA 073 Cortados y graveras del Jarama (2), E: IBA 393 Torrejón de Velasco-secanos de Valdemoro (1), F: IBA 072 Carrizales y sotos de Aranjuez (1).

Cereal fields are harvested during late June to early July. Stubbles and fallows are also used for sheep grazing. These areas hold populations of threatened bird species such as great bustard *Otis tarda* (ca. 1 500 individuals, Alonso *et al.*, 2003), little bustard *Tetrax tetrax* (ca. 2 600 individuals, García de la Morena *et al.*, 2006) pin-tailed and black-bellied sandgrouses *Pterocles alchata* and *P. orientalis*

(ca. 112 and 100 individuals, respectively, Suárez *et al.*, 2006), and montagu's harrier *Circus pygargus* (ca. 100 pairs, Arroyo & García, 2007).

Carcass detection and removal by scavengers

Between November 2007 and August 2008 we carried out five carcass disappearance trials, respectively in November, December, February, April and August. Each trial started by placing the bird carcasses on the ground under a power line (20 and 5 carcasses/ km for November and the rest of the months, respectively). The line was then surveyed four times through the month following placement (on days 2, 7, 22 and 28; in December it was not possible to carry out the survey on day 28 due to unfavourable weather conditions). We searched at uneven intervals because most of the disappearances are known to occur during the first days after the casualties (e.g. Balcomb, 1986; Prosser *et al.*, 2008). With the aid of the GPS we went to each site where we had placed a carcass and looked for it or its remains, recording any track or trace left by scavengers. On the last survey day of each trial we removed all carcass remains.

In total, 522 carcasses were placed at 0-20 m from the line beneath the central conductor wire of the power line to simulate natural collisions (Henderson, Langston & Clark, 1996; Janss, 2000). One hundred and thirty of these carcasses were female common pheasants (*Phasianus colchicus*), 130 red-legged partridges (A*lectoris rufa*), 130 common quail (*Coturnix coturnix*), and 132 halves of common quail carcasses. We chose these species because they are found in the study area; pheasants were intended to represent a bird of similar size and plumage to great bustards, the largest species, while common quail halves should represent small passerines. Using four size classes (pheasants were Large, partridges were Medium, quail were Small, and half-quail were very Small) allowed us to explore the effect of carcass size on removal probability. All carcasses used were from wild birds hunted and later sold for human consumption, thus they were free from the smell characteristic of poultry farm birds, which might have influenced the removal rate by scavengers (Bevanger, 1999). For this reason we preferred wild common quail halves to any other small farm bird like small chicken or ducks. Significant weight differences existed between the four size category used ($p < 0.001$ in all cases; common pheasants: 1008.9 g ($\pm$ 125), n = 20; red-legged

partridges: 406.3 g (± 42.0), n = 25; common quail: 109.5 g (± 14.2), n = 25; common quail halves: 54.1 g (± 6.3), n = 24). All carcasses were aired in a ventilated and cold room for 24 h prior to placing it under the power line to eliminate as much as possible any artificial smell remains but avoiding decomposition due to temperature, which may reduce the attractiveness to vertebrate scavengers.

We considered that a carcass had been detected by a scavenger when it had been moved from the initial location, partially eaten, or completely removed. A carcass disappeared when the remains found were less than 5 feathers, because a very low number of feathers found during searches for collision casualties cannot be interpreted as a collision, as these few feathers could have been lost by a bird during preening, moulting or fighting (e.g. Bevanger, 1999). We searched for carcasses up to 30 m away from the initial location to account for possible dragging of the carcass by scavengers. To look at possible differences in removal rate due to changes in density of carcasses (see e.g. Linz *et al.*, 1991; Wobeser & Wobeser, 1992; Ward *et al.*, 2006), we placed them at respectively 50 m- (20 carcasses/ km) and 200 m-intervals (5 carcasses/ km) in two winter trials. As no differences were found, in all other trials we placed carcases at 200 m-intervals. The placement order of the four size classes was random. For each carcass placed we recorded UTM coordinates with GPS (Garmin, ± 3 m error), and vegetation cover and average height (estimated visually in a circle of 3 m radius around the carcass). Before placing the carcass we made a cut on its ventral side to simulate the injury caused by the collision with the cable and to avoid differences respect to the smallest size (common quail halves).

Carcass detection by observers

We explored the influence of the observer's experience on carcass detectability during the first two experiments (141 carcasses). The experience was defined as the total kilometres surveyed under power lines by each observer before the present study was carried out. Four observers different from those who had placed the carcasses surveyed the power lines searching for remains. Each of these surveys was conducted by two observers, one after another separated by ca. 50 m,

walking at a slow, regular pace and parallel to the wires of power lines at a distance no more than 15 m of the central conductor wire. The visibility was good along all the power line corridors due to low height of the vegetation, so the observer could find all the remains to a distance up to 50 m. The first observer searched for remains without knowing where the carcasses had been placed; the second walked behind him recording both the remains discovered and those not found by the first observer.

Statistical analyses

To establish the factors influencing the carcass disappearance rate we used a generalized linear model with a binary response (carcass or its remains disappeared vs. present on day 28 after placement). As factors we included each one of the power lines, month, carcass size, and vegetation cover and height, after appropriate transformations for vegetation variables -natural logarithm (height+1) and arcsine ($\sqrt{\text{cover}}$)- to attain equal variance and normality (Sokal & Rohlf ,1987; Fowler, Cohen & Jarvis, 1998). To explore the relative importance of each explanatory variable, we used the corrected Akaike's information criterion (ΔAICc<2) to select the best models from a set of candidate models with different combinations of predictor variables (Anderson & Burnham, 1999) and interactions among them.

Once the relevant factors were identified we performed univariate analyses to further explore their influence on the carcass disappearance rate. We used some non parametric tests because we investigated several questions about the different power lines (11) and months (5) that were considered as independent experiments. (i) Mann-Whitney *U*-tests to investigate the importance of the carcass density, by comparing the number of carcasses disappeared between high density - November experiment- and low density -all other experiments-; (ii) Chi-squared tests to search for differences among power lines due to variable carcass density; (iii) Kruskal-Wallis tests to check for seasonal differences between experiments carried out on different months; (iv) Kruskal-Wallis and Mann-Whitney *U*-tests to explore differences due to carcass size; (v) Chi-squared tests with Yates correction when necessary (Fowler *et al.*, 1998) to look at removal rate differences between months or power lines. Finally, to describe the removal rate as a function of

carcass size we adjusted a logarithmic equation to disappearance data for each carcass size.

To investigate the effect of the observer's experience on carcass detectability we performed a second generalized linear model with logit link function and a binary response (carcass or remains found vs. not found), using as factors the observer, carcass size, vegetation height and vegetation cover. We applied the same variable transformations and model selection criteria used in the previous analysis. Also, we carried out univariate analyses to explore (i) whether large carcasses were detected with higher probability than small ones, and (ii) differences between observers in their ability to find the remains which could be attributed to their previous experience. As an estimate of experience we used the kilometres of power line each observer had patrolled looking for collision casualties prior to this study. We finally adjusted logarithmic equations to detectability data for each observer.

RESULTS

Carcass detection and removal by scavengers

On the first survey, two days after leaving the carcasses under the power lines, 67.2% of them had been detected by scavengers, with no differences among bird sizes (χ^2 = 0.94, d.f. = 3, P < 0.82). Detection rate increased to 93.7% during the second survey (day 7), with no size differences (χ^2 = 0.12, d.f = 3, P < 0.99), and to 99.8% and 100%, respectively for the third and fourth surveys (days 22 and 28).

The accumulated number of carcasses removed by scavengers increased from the day they were placed following a logarithmic function (Figure 2). On day two, 32% of the carcasses had already disappeared. An additional 20% of the carcasses disappeared between days 2 and 7, a further 16% between days 7 and 22, and only 3% between days 22 and 28. Disappearance rates for each survey date did not change between experiments carried out on different months (P > 0.08 in all cases). On day 28 after placement of the carcasses under the power lines, 71.5% of the initial sample had disappeared. This carcass disappearance rate

was not influenced by carcass density, either considering all power lines together in a sample (Z = 1.35, $P < 0.18$, November vs. all other months; $\chi^2 = 0.6625$, d.f. = 1, $P < 0.42$ between two winter tests -November and February- to control for a possible seasonal effect), or testing each power line separately ($P > 0.18$ in all cases).
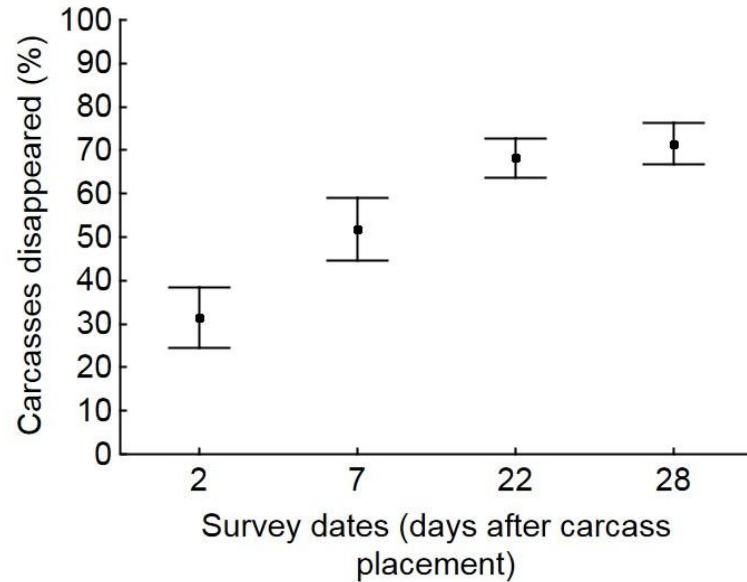
Figure 2. Accumulative percent of carcasses disappeared on the different survey dates (= day after carcasses were placed under the power line). Means and SD are given.

The result of the generalized model showed that carcass disappearance on day 28 was influenced by carcass size (at higher speed for smaller carcasses) and power line, with no significant effects of other variables or interactions among them (Table 1).

Table 1. Results of the generalized linear model for carcass disappearance on the last survey date (day 28 after placing carcasses).

Variable	Partial deviance	P
Carcass size	76.43	0.001
Power line	28.17	0.001
Month	4.34	0.226
Month*Carcass size	2.37	0.498
Vegetation height	0.00	0.961
Vegetation cover	0.10	0.749

There were three power lines where disappearance rates differed from the rest: Belvis–Cobeña and El Casar–La Cueva, where disappearance rate was

respectively 23% and 19% lower than average), and Pinto–San Martín de la Vega, where it was 20% higher. The model was highly significant ($\chi 2$ = 133.016, d.f. = 19, P < 0.001), explaining 39.5% of the total deviance. Carcass size was included in the first eight models selected as the best subsets (all eight were highly significant, P < 0.001, Table 2), confirming its higher relevance as compared to power line (included in models 1-4 and 9-11). Vegetation height and cover appeared respectively in models 2 and 3, as well as in various successive models, all of them with ΔAICc > 2 (Table 2).

Table 2. Models selected as best significant subsets by the generalized linear model for carcass disappearance (see Table 1), ranked according to ΔAICc.

Nº Model	AICc	ΔAICc	w_i[a]	k[b]	P[c]
1 Carcass size-Power line	426.78	0	0.534	13	0.001
2 Carcass size-Power line-Vegetation height	428.88	2.10	0.187	14	0.001
3 Carcass size-Power line-Vegetation cover	428.90	2.12	0.185	14	0.001
4 Carcass size-Power line-Vegetation height-Vegetation cover	430.99	4.21	0.065	15	0.001
5 Carcass size	433.86	7.08	0.015	3	0.001
6 Carcass size-Vegetation height	435.84	9.06	0.006	4	0.001
7 Carcass size-Vegetation cover	435.89	9.11	0.006	4	0.001
8 Carcass size-Vegetation height-Vegetation cover	437.74	10.96	0.002	5	0.001
9 Power line	528.95	102.17	0.000	12	0.027
10 Power line-Vegetation height	531.01	104.23	0.000	13	0.041
11 Power line-Vegetation cover	531.05	104.27	0.000	13	0.041

[a] model weight
[b] number of parameters
[c] significance of the model

The function describing the disappearance rate through the first month for each carcass size is shown in Figure 3. On survey date 28, 42.5% of large, 62.1% of medium-sized, 86.9% of small, and 93% of very small carcasses had disappeared, with significant differences among these values (H 3, 16 = 13.08, P < 0.005). The differences were significant between large and medium (Z = -2.31, P < 0.021), and between medium and small (Z = -2.32, P < 0.020), but not between small and very small carcasses (Z = -1.15, P < 0.25). Disappearance rates for each carcass size did not change with carcass density (P > 0.54, P > 0.47, P > 0.46, and P > 0.50, respectively from large to very small), or power line ($\chi 2$ = 0.28 < P 0.99). Using the

weights of the four size classes we obtained an equation predicting the disappearance rate at 28 days as a function of weight (Figure 4).
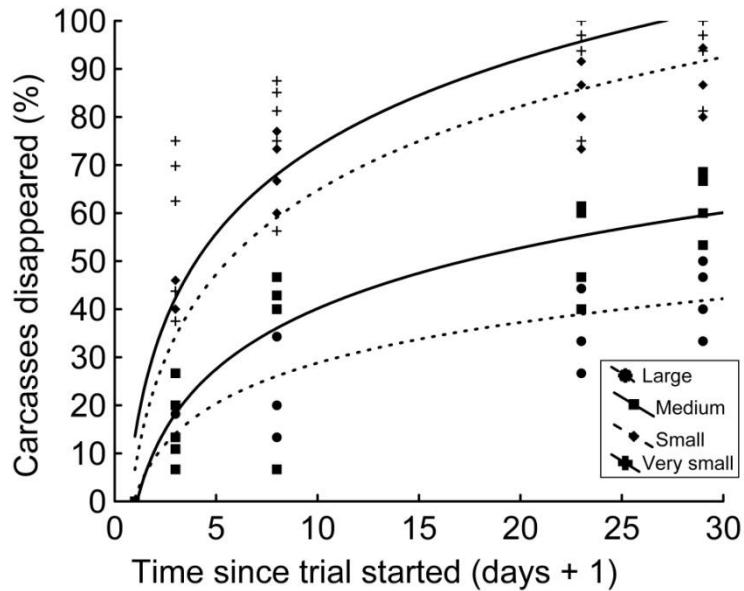
Figure 3. Accumulative percent of carcasses of each size disappeared through the four surveys (days 2, 7, 22 and 28; survey dates were transformed as x=day+1). For each carcass size, five data corresponding to the five trials conducted on different months are represented (November, December, February, April and August; in December it was not possible to carry out the survey on day 28 due to unfavourable weather conditions). The curves represent the logarithmic models that fitted best to these monthly disappearance figures. Large size: y = 0.744+28.063*log10(x) (r = 0.83); Medium size: y = -1.751+41.880*log10(x) (r = 0.88); Small size: y = -6.623+58.111*log10(x) (r = 0.84); Very small size: y = 13.538+60.342*log10(x) (r = 0.75). *P* < 0.001 in all cases.
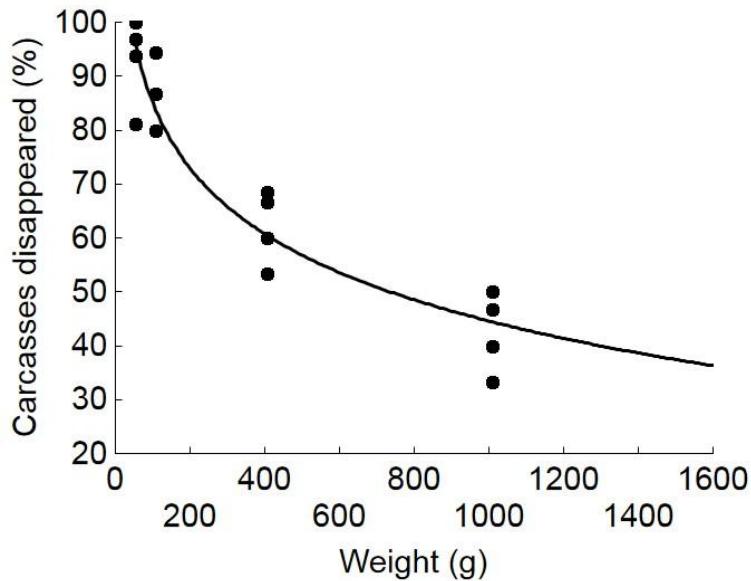
Figure 4. Percent carcasses disappeared on the last survey (day 28) for each bird weight class. Black dots are the values for the four trials (November, February, April and August). The curve represents the logarithmic equation adjusted to these values: $y = 166.295 - 40.567 * \log10(x)$ ($r = 0.93$, $P < 0.001$).

Carcass detection by observers

On average, an observer discovered 53% of the carcasses present. However, there were significant differences in their ability to find the remains ($\chi^2 = 3.88$, d.f. = 1, $P < 0.05$; observers A, B, C and D found respectively 25%, 57.1%, 68.4% and 70.4% of the carcasses). The generalized model showed that carcass detectability was influenced by carcass size and observer, with no significant effect of vegetation height or cover and their interaction (Table 3). The model was highly significant ($\chi^2 = 38.56$, d.f. = 7, $P < 0.001$), explaining 20.0% of the total deviance. Large carcasses were detected in higher proportion (71.7 %) than other sizes (respectively, 55.8 %, 32.1 %, and 33.3 % for medium-sized, small, and very small carcasses, $\chi^2 = .03$, d.f. = 1, $P < .05$), with no differences among medium to very small sizes ($P > 0.08$ in all cases). Fifteen significant candidate models were obtained, of which the first two showed ΔAICc < 2 and included observer (not in model 2), carcass size and vegetation height (Table 4). Using the kilometres of power line surveyed by each observer prior to this study as an index of his experience in detecting carcasses, this factor explained 92% of the variation of the detection rate (Figure 5).

Table 3. Results of the generalized linear model for carcass detectability.

Variable	Partial deviance	P
Carcass size	16.42	0.001
Observer	8.38	0.039
Vegetation height	2.26	0.133
Vegetation cover	0.00	0.965
Vegetation height*Vegetation cover	1.87	0.140

Table 4. Models selected as best significant subsets by the generalized linear model for carcass detection rate (see Table 1), ranked according to ΔAICc.

Nº Model	AICc	ΔAICc	w_i[a]	k[b]	P[c]
1 Observer-Carcass size-Vegetation height	171.18	0	0.431	7	0.001
2 Carcass size-Vegetation height	173.07	1.89	0.166	4	0.001
3 Observer-Carcass size-Vegetation height-Vegetation cover	173.42	2.25	0.140	8	0.001
4 Observer-Carcass size-Vegetation cover	173.43	2.26	0.140	7	0.001
5 Carcass size-Vegetation height-Vegetation cover	175.15	3.98	0.059	5	0.001
6 Observer-Carcass size	176.15	4.98	0.036	6	0.001
7 Carcass size-Vegetation cover	176.84	5.67	0.025	4	0.001
8 Observer-Vegetation height	185.42	14.24	0.000	6	0.001
9 Carcass size	185.75	14.58	0.000	3	0.001
10 Observer-Vegetation cover	186.24	15.07	0.000	6	0.001
11 Observer	186.99	15.82	0.000	5	0.001
12 Observer-Vegetation height-Vegetation cover	187.60	16.42	0.000	7	0.001
13 Vegetation height	188.60	17.42	0.000	3	0.001
14 Vegetation height-Vegetation cover	190.58	19.41	0.000	4	0.006
15 Vegetation cover	190.64	19.47	0.000	3	0.004

[a] model weight
[b] number of parameters
[c] significance of the model
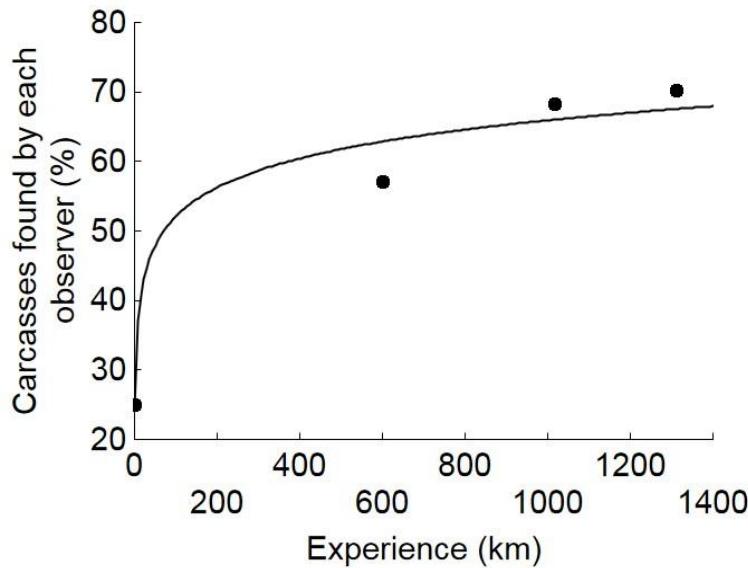
Figure 5. Detection ability of the four observers participating in the detectability trial (black dots), as a function of their experience (defined as the number of kilometres of power line surveyed prior to the present study). The curve represents the equation adjusted to the four detection ability values, $y = 24.461 + 13.827 \times \log10(x)$ ($r = 0.961$, $P < 0.04$).

DISCUSSION

Our results indicate that removal of carcasses by scavengers reduced the number of dead birds placed initially under power lines. The number of carcasses present followed a logarithmically decreasing trend through the days following trial start. Second, searcher efficiency biased the number of carcasses low to a lower level by a variable extent, depending on previous personal training. Third, these two sources of bias increased with decreasing carcass size, and removal rate was also site-dependent. Fourth, the corresponding corrections should be taken into account when using carcass surveys to calculate bird mortality estimates due to electrocution or collision at power lines. Below we discuss these results in detail.

These conclusions can be drawn from our study, in spite of the following methodological limitations which could have affected the scavenging rates obtained. For example, our presence in the area and handling of the carcasses when placing them may have either attracted or deterred scavengers. Scavengers could have followed human trails to carcasses or, alternatively, shy species might have avoided carcasses or sites tainted with human scent (Wobeser & Wobeser,

1992). We believe, however, that these effects were negligible because in our study area scavengers are likely to be used to human presence due to the frequent occurrence of human activities such as farming, sheepherding and hunting. We tried to minimize other possible sources of error based on carcass odour or conspicuousness. The results of previous studies have suggested that brighter-coloured corpses may be more conspicuous and easier to be detected by aerial scavengers (e.g. Balcomb, 1986; Prosser *et al.*, 2008). This would however not influence removal rates by mammalian scavengers, which mostly search by scent and are nocturnal. More frequently, authors have drawn attention to the removal rates between wild bird carcasses and those of artificially reared species (Balcomb, 1986; Young *et al.*, 2003; Prosser *et al.*, 2008). We used exclusively wild birds shot by hunters to minimize these odour-based effects. Moreover, we left corpses one day aired in a ventilated and cold room before placing them to eliminate any scent from handling by hunters and suppliers. Also, the species we used belonged to the local fauna and were similar in plumage colour and pattern to most other steppe-birds living in the study area. Another source of variation in removal rate may be the carcass density. Obviously, in carcass removal trials carcass density is higher than in most natural events, in order to make searches and calculations feasible within reasonable time and space limits. Some authors have suggested that greater than normal carcass abundance may attract scavengers and either increase removal rate (Bevanger *et al.*, 1994), decrease it due to satiation (Linz *et al.*, 1991), or produce no observable effects (Wobeser & Wobeser, 1992; Prosser *et al.* 2008). Another studies carried out in the same power lines by the authors showed that around 8 wild dead birds/ km were found under them during one year sampling (one each month) so, if we consider that many of the collided or electrocuted birds may have been moved by scavengers or not found by the observers (as we have demonstrated in the present work), we can assume that we have not significantly increased the density of dead birds with respect to normal casualties. But to check for this possibility in this experiment, we compared our standard density with a four-fold density, and found no differences in removal rate.

Carcasses were removed by scavengers with highest intensity immediately after placement. Later, removal rate decreased regularly through a period varying between some days and several weeks. The accumulative disappearance curves

best fitting the data were logarithmic and similar in shape for all four size classes tested, but smaller carcasses disappeared earlier and in higher proportion than larger ones. Our results show that removal rate increased with decreasing carcass size, except for the two smallest size classes which were removed at similar rates. These smaller carcasses were most frequently removed without leaving any remains (66.7% small and 85.7% very small carcasses removed on day 2), in contrast to big corpses which were normally partially eaten on the spot (on day 2, 78.8% medium and 73.6% large corpses; all size differences significant, $P < 0.02$). Remains of larger corpses were easily recognized through the whole series of search surveys, most often ending up as a pile of feathers that usually remained for several weeks on the spot, indicating past scavenger activity. These facts suggest that a wider spectrum of scavenger species were able to feed on and remove corpses below a certain size, whereas potential predators able to remove larger carcasses at once were much scarcer, and these large corpses were discovered and as a rule incompletely devoured by the same scavengers as those feeding on the smaller corpses. Common scavengers in our study area include mammals like fox (*Vulpes vulpes*), feral dogs (*Canis familiaris*), feral cats (*Felis silvestris catus*) or black rat (*Rattus rattus*), and birds such as black and red kites (*Milvus migrans* and *M. milvus*), corvids like magpies (*Pica pica*), jackdaws (*Corvus monedula*), ravens (*C. corax*), white storks (*Ciconia ciconia*), and black-headed and black-backed Gulls (*Larus ridibundus* and *L. fuscus*). The fact that we didn't find differences among carcass sizes in the scavenger detection rate (which includes both disappeared and partially eaten carcasses) indicates that corpses were found opportunistically, and not due to their visibility. This suggests that the most frequent scavengers in our study area were probably mammals, which mostly hunt by scent (see also Kostecke, Linz & Bleier, 2001 for the same interpretation based on results confirmed through photographic evidence). Smallwood *et al.*, (2008) found 74% and 63% of the carcasses respectively detected and removed by mammals, although in that study differences among carcass sizes were found. However, identification of all scavenger species and their relative contribution to the disappearance of carcasses was not among our objectives.

Previous studies have also found decelerating removal rates (Balcomb, 1986; Ward *et al.*, 2006), and very high initial removal rates among smaller

carcasses, most of which disappeared within the first days (Heijnis, 1980; Wobeser & Wobeser, 1992; Prosser *et al.*, 2008). However, few of these studies followed carcasses for more than a week, which renders estimates of the eventual fate of certain carcasses difficult, particularly of the larger ones which usually survive longer. In our study we surveyed the power lines through four weeks after placement, because one of our main objectives was to determine the frequency with which carcass searches should be conducted to determine fatalities at power lines. Although most mortality studies at power lines are based on weekly to monthly survey frequencies, such periodicity is usually fixed without a well-founded basis. The disappearance curves obtained in our study through a month for various bird sizes offers the opportunity to determine an acceptable search frequency, depending on the bird species for which removal rates are required. An interesting result not found in most previous studies was that for all four carcass sizes tested, further removals were recorded even over 20 days after placing the corpses.

The second factor influencing removal rate was the power line. No significant effects were found from other variables such as season or vegetation structure, suggesting a relatively uniform scavenger pressure through the year and among different substrate types. Changes in scavenger density have been suggested to be the main reason for the differences in removal rate found among sites (Kostecke *et al.*, 2001), seasons (Bevanger *et al.*, 1994; Linz *et al.*, 1991; Johnson *et al.*, 2003; Prosser *et al.*, 2008), or areas with different vegetation structure (Bevanger *et al.*,1994; Bevanger, 1995; Siriwardena *et al.*, 2007). In our study, only three power lines showed unusual removal rates. The line Pinto - San Martín de la Vega was close to a huge rubbish dump, where large numbers of black kites and white storks are found in spring and summer, and black-headed and lesser black-backed gulls aggregate by thousands, mainly in winter. Individuals of all these species have wide home ranges and could have easily contributed to the higher carcass removal rate recorded at this power line. The two power lines with lowest removal rates were located in close proximity to villages, which might have determined a lower density of scavengers and, therefore, a lower removal rate. However, the purpose of our study was only to explore the relative amount of local or seasonal differences and their effect on removal rate, nor to investigate the

causes of such differences. Based on the significant differences found in three of eleven lines, we conclude that scavenger rates are probably site-dependent in most cases. Moreover, although seasonal differences in removal rate did not reach statistical significance in our study, the range of values obtained for different months was quite wide, which suggests that seasonal variation could be an important factor to be considered in future studies. A similar conclusion can be drawn for vegetation structure, which did not appear to significantly affect removal rate, but appeared on some of the candidate models selected in our analyses. Overall, this suggests that local, seasonal, and other differences due to vegetation structure, may affect scavenger removal rate to a variable extent, and therefore the figures given in the present study should be taken with care. For example, a more dense, diverse or higher vegetation could be an influential variable in studies focusing on small birds. The correction indices derived from our trials could probably be applied to estimate power line-caused mortality in similar habitats within the Mediterranean region, being less useful for areas differing much in geographic location, habitat structure or scavenger community. Studies similar to the present one should be conducted in areas with completely different climatic conditions, i.e. where the ground is covered with snow through several months in winter, or the vegetation and habitat structure are quite different, in order to check the importance of weather and vegetation variables and obtain more reliable correction factors.

Finally, the four observers participating in this study differed notably in their ability to find carcasses (25-70.4%). A similar range in detectability values has also been reported in previous studies (e.g., 35-85% in Morrison's 2002 review). Lower detection rates have been attributed to a higher (Philibert, Wobeser & Clark, 1993) or denser (Wobeser & Wobeser, 1992) vegetation. In our farmland study area, changes in vegetation structure were probably not enough to determine significant variations in detectability. The two factors that we found to influence detectability were carcass size and previous experience of the observer. Larger carcasses were detected in higher proportion than smaller ones, as reported in Siriwardena's (2007) review of wind turbine-caused mortality. The correlation found in our study between detection rate and previous experience of the observer specifically conducting these kinds of searches at power lines is an

important new result that highlights the importance of a training period for field workers participating in carcass searches intended to estimate mortality rates at power lines. We cannot exclude that other factors, e.g. personal motivation may influence the searching detection rate. Finally, the results should be interpreted with caution, due to the small number of observers that have participated in the experiment.

CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Carcass counts at power lines will notably underestimate the number of bird casualties, the bias being higher in smaller birds. Mortality estimates should incorporate correction factors based on scavenging rates and observer efficiency. Conservation authorities and power line operators should be aware of these bias sources and adjust past and future estimates before using them to assess power line-caused bird mortality. Scavenger removal rates differed to a great extent with carcass size, being much higher for small birds. A high percent of these small carcasses had disappeared two days after placement, and ca. 90% after two weeks. This indicates that fortnightly to monthly search frequencies may be adequate to detect casualties of medium to large-sized species, but are insufficient in the case of smaller species. For these, a higher search frequency is recommended, in order to reduce the uncertainty interval implicit in extrapolations from equations like those presented here. Although site-related and seasonal differences found in our study did not reach statistical significance, the range of values obtained for a sample of 55 surveys (5 months × 11 power lines) was considerable. This suggests that, if precise mortality estimates are required, scavenger removal trials should be carried out simultaneously with searches aiming to estimate collision mortality. We recommend carrying out such complementary removal trials whenever possible. Alternatively, the equations presented here may be used to obtain mortality estimates in Mediterranean farmland. Figures may vary substantially between this and other farmland habitats at different latitudes. Therefore, similar studies are needed in these habitats to evaluate the effects of various bias sources affecting scavenger removal rates there. Finally, all personnel participating in

carcass searches should be previously trained in this task, in order to minimize detection errors due to low experience.

ACKNOWLEDGEMENTS

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CAPÍTULO 2

Este capítulo reproduce íntegramente el siguiente artículo:

Barrientos, R., Ponce, C., Palacín, C., Martín, C.A., Martín, B. & Alonso, J.C. 2012. Wire Marking Results in a Small but Significant Reduction in Avian Mortality at Power Lines: A BACI Designed Study. *Plos One 7*(3): e32569. doi:10.1371/journal.pone.0032569.

CAPÍTULO 2

Wire Marking Results in a Small but Significant Reduction in Avian Mortality at Power Lines: A BACI Designed Study

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ABSTRACT

Background: Collision with electric power lines is a conservation problem for many bird species. Although the implementation of flight diverters is rapidly increasing, few well-designed studies supporting the effectiveness of this costly conservation measure have been published.

Methodology/Principal Findings: We provide information on the largest worldwide marking experiment to date, including carcass searches at 35 (15 experimental, 20 control) power lines totalling 72.5 km, at both transmission (220 kV) and distribution (15 kV-45 kV) lines. We found carcasses of 45 species, 19 of conservation concern. Numbers of carcasses found were corrected to account for carcass losses due to removal by scavengers or being overlooked by researchers, resulting in an estimated collision rate of 8.2 collisions per km per month. We observed a small (9.6%) but significant decrease in the number of casualties after line marking compared to before line marking in experimental lines. This was not observed in control lines. We found no influence of either marker size (large *vs*. small spirals, sample of distribution lines only) or power line type (transmission *vs*. distribution, sample of large spirals only) on the collision rate when we analyzed all species together. However, great bustard mortality was slightly lower when lines were marked with large spirals and in transmission lines after marking.

***Conclusions*:** Our results confirm the overall effectiveness of wire marking as a way to reduce, but not eliminate, bird collisions with power lines. If raw field data are not corrected by carcass losses due to scavengers and missed observations, findings may be biased. The high cost of this conservation measure suggests a need for more studies to improve its application, including wire marking with non-visual devices. Our findings suggest that different species may respond differently to marking, implying that species-specific patterns should be explored, at least for species of conservation concern.

INTRODUCTION

Bird collisions with electric power lines have raised conservation concerns since the early 1900s, but it was not until the 1970s that biologists and engineers began to realize the extent of this problem [1,2]. Today the number of power lines is increasing worldwide at an annual rate of approximately 5% [3]. Mortality from collisions with power lines and other electric utility structures has been documented for some 350 bird species [4]. However, until a cumulative impacts assessment of power line mortality is conducted, the real level of mortality will remain uncertain [5]. Only some crude estimates of the importance of the problem, all of them based on extrapolations, are available. For example, in the Netherlands it has been found that bird collisions with power lines may cause one million deaths per year [6]. In the United States, [5] it is estimated that power lines may kill up to 175 million birds annually, and it is estimated that bird collisions with power structures, including transmission (≥70 kV, usually with ground-wire and wires at more than one height) and distribution (<70 kV, commonly without ground-wire and all the wires at the same height) lines, could approach one billion avian fatalities per year worldwide [7]. Fortunately, these values are probably overestimated since most of the studies are usually carried out on power lines that cause an important number of fatalities. Nevertheless, these figures allow conservationists to speculate that mortality due to collisions with power lines represents a serious threat for population viability in many species, at least in those that undergo higher collision risks, and that this threat is not equal for all species. Indeed, birds with low manoeuvrability, i.e., those with high wing loading and low aspect, such as bustards, pelicans, waterfowl, cranes, storks, and grouse, are among the species most likely to collide with power lines [2,8]. Species with narrow visual fields are also at high collision risk as they do not see the wires [9,10]. Despite this potentially important conservation problem, few studies have analyzed in detail how these losses affect population trends. For instance, it has been estimated that collision-related losses might equal up to 90% of the annual number of grouse harvested by hunting in Norway [11]. Based on ring-recovery data [12], it has been assessed that 25% of juveniles and 6% of adult white storks

(*Ciconia ciconia*) die annually in Switzerland due to power lines (although these data also include electrocutions). It has also been estimated that 30% of Denham's bustards (*Neotis denhami*) die annually by collisions with power lines in South Africa [13].

Researchers and managers have used several methods to reduce collisions, including the removal of the static wire [14,15]. However, the most popular measure has been the attachment of spirals, plates, swivels, or spheres (collectively known as *bird flight diverters*) to the static wire in order to increase visibility [3,16,17,18]. While a recent review concluded that marking static wires reduces the overall number of bird casualties at power lines, it also called attention to the fact that there are a surprisingly small number of well-designed, peer-reviewed studies to support this [19]. Furthermore, there remain many gaps in the research in this area, with several important details still unresolved; for example, the comparative effectiveness of various currently available marker types [19]. To confirm diverter effectiveness, and to study all details of this conservation measure in depth is especially important because despite the high costs of wire marking (e.g., 1,100-2,600 US$ per marked kilometre in South Africa, [20]; 6,000€ in Spain; [21]), the application of this conservation measure is rapidly increasing worldwide.

As stated above, it has been shown that the presence of flight diverters was associated with a decrease in bird collisions [19]. However, the large differences in wire-marking techniques constrained the ability to evaluate potential differences among methods (e.g., different performance based on diverter traits) in that review. To complement such an approach, in the present study we designed the largest field experiment to date, to investigate: (i) the effectiveness of wire marking in reducing collisions; and the roles of (ii) power line type (transmission *vs.* distribution), and (iii) spiral size on marking effectiveness. We expected that: (i) the attachment of spirals would reduce bird mortality [19]; (ii) the effectiveness of marking would be higher in transmission lines because power line type influences the frequency of reactions to marked spans [22]. Morkill & Anderson [22] found that whooping cranes (*Grus americana*) reacted more than expected to transmission lines (345 kV, 27 m high) whereas the opposite was true in

distribution lines (69 kV, 12 m high). It is worth noting that transmission lines in our study accumulate a larger number of collisions of those groups of birds especially prone to collision, such as bustards, storks or waterfowl (see below) compared to distribution lines. Therefore, the improvement margin once spirals are attached is greater in transmission lines; and, (iii) larger spirals may be more effective in increasing the visibility of wires [23,24], reducing collisions to a larger extent.

METHODS

Study area

The study was conducted in five important bird areas (IBAs) in central Spain (see [25] for details), which are also the main dry cereal farmland areas in the Madrid region. The terrain is flat to slightly undulating, with a mean elevation of c. 750 m a.s.l. These areas are primarily dedicated to cereal cultivation (mainly wheat *Triticum aestivum* and barley *Hordeum* spp.), with minor fields of legumes *Vicia* spp., grapevines *Vitis vinifera* and olive *Olea europaea* groves. Most cereal is grown in a traditional 2-year rotation system that creates a dynamic mosaic of ploughed, cereal and stubble patches over the region. Small patches of natural vegetation (holm oaks *Quercus ilex*, and scrubland of *Retama* spp. and *Thymus* spp.) remain dispersed across the cereal matrix. Cereal fields are harvested in late June to early July. Stubbles and fallows are also used for sheep grazing [26].

Study species

We considered all birds that we found dead under the power lines in the study area. We discarded the dead birds found beside poles whose cause of death could be attributed to electrocution. However, since not all species have the same collision risk [2,8,9], it is worth noting that the study area holds significant populations of threatened species which are prone to high collision rates due to their low manoeuvrability, high speed flight and/or poor vision [2,8,9], such as the great bustard *Otis tarda* (c. 1500 individuals; [27]), little bustard *Tetrax tetrax* (c.

2600 individuals; [28]), pin-tailed sandgrouse *Pterocles alchata* and black-bellied sandgrouse *P. orientalis* (c. 150 and 200 individuals, respectively, [29]).

Study design and power line monitoring

The study was carried out using a before-after-control-impact (BACI) design, i.e. monitoring power lines before and after the placement of spirals, combined with the use of controls during similar time intervals. Between August 2001 and December 2010 we surveyed bird collisions monthly at 22 different power lines, 7 of them transmission (220 kV) and 15 distribution (15 kV-45 kV) lines, totalling 16.1 and 27.0 km, respectively (Table 1). Fifteen of these lines were our *experimental* lines, i.e. to which spirals were attached. These were monitored once per month for two complete years (one year *before* and one year *after* wire marking). Another 7 lines to which no spirals were attached were used as *control* lines and were monitored also once per month for two complete years. Because no more non-marked control lines were available, in addition to these 7 control lines we also used as controls the second of 10 two-year and the third of 3 three-year surveys carried out at experimental lines once spirals were attached to them (Table 1). These surveys can be considered as *controls* since once the line was marked no changes occurred in the factor presence/absence of spirals and thus no changes were expected between years in the variable under study, i.e. collision rate. The resulting number of power lines (35) and the total length surveyed monthly (72.5 km) for all study years make our study both the most detailed and that with the largest number of power lines monitored to date (for instance, the mean number of power lines per study was 1.9 in a recent review, see Appendix S2 in [19]).

Table 1. Power line name, type of line (transmission or distribution), design (experimental or control) and number of years monitored after spiral attachment.

Power line	Type	Length (km)	Design	Times after
Aranjuez E-O	Distribution	2.0	Control	One
Aranjuez N-S I	Transmission	2.0	Experimental	One
Aranjuez N-S II	Transmission	4.1	Experimental	One
Belvis-Cobeña	Transmission	3.0	Experimental	Three
Camarma-Fresno	Distribution	2.0	Experimental	Two
Camarma-Meco	Transmission	1.6	Experimental	Two
Camarma-Torote	Transmission	2.1	Experimental	Three
Campo Real-Valdilecha	Distribution	3.2	Experimental	Two
Daganzo-Alcalá	Distribution	0.9	Control	One
Daganzo-Fresno Rio	Distribution	1.1	Control	One
Daganzo-Torote	Transmission	1.8	Experimental	Three
El Colegio	Distribution	3.0	Experimental	Two
La Cueva-El Casar	Distribution	1.5	Control	One
Mesones	Distribution	2.0	Control	One
Pinto	Transmission	1.5	Experimental	Two
Pozuelo-Valdilecha	Distribution	2.6	Experimental	Two
Quer	Distribution	1.4	Experimental	One
San Martín de la Vega	Distribution	1.7	Experimental	Two
Valdepiélagos-Talamanca I	Distribution	2.2	Experimental	One
Valdepiélagos-Talamanca II	Distribution	0.5	Control	One
Valdetorres-La Jara	Distribution	1.4	Control	One
Villanueva-Quer	Distribution	1.5	Experimental	One

One month before the beginning of each monitoring year we removed all carcasses under the power line. Each monthly search for bird carcasses was carried out by one observer walking at a slow, regular pace parallel to the wires but making zigzags to reasonably visually cover a 25 m band at each side of the vertical of the central conductor wire. The observer surveyed first one side along the line (e.g. the 25 m band on the right side), and then he/she returned to the starting point surveying the other side (25 m band on the left side). All remains found were identified to the species level and removed to avoid double counts. When the species was unknown (<2% of the cases), the carcass was assigned to one of the four sizes considered (see below). We recorded a carcass when the remains found consisted of more than five feathers in a square meter, because a smaller number of feathers cannot safely be interpreted as a collision, since they

could have been lost by a bird during preening, moulting or fighting [30]. Carcass searches were not performed in June because crop height may lead to unrealistically low carcass detection figures. July surveys were always carried out after cereal harvesting. However, it is worth noting that in our rather structurally-homogeneous study area, there was no relationship between vegetation height or cover and carcass detection rates [25].

Potential detection biases such as site- or year-dependent carcass removal by scavengers or variation in carcass detection due to habitat heterogeneity are minimized in our study, since we used a BACI design combined with the use of control power lines at the same time intervals. Furthermore, potential outbreaks in scavenger populations are unexpected because predator control is widespread in our study region [31]. However, since monthly search frequencies may be adequate to detect medium- to large-sized corpses, but are insufficient for smaller birds, we used equations from [25] to adjust our mortality estimates in relation to search periodicity and carcass size (Table 2), because both can influence mortality estimates. The correction of field data is important because larger carcasses are detected by researchers more easily than smaller ones, and because the longer time elapsed between consecutive searches and the smaller the size of the carcasses, the larger the effect of scavengers on corpse disappearance [25]. Ideally, surveys to evaluate carcass losses should be carried out in each study area before undertaking further mortality studies [25], because detection rates can differ among study areas (e.g., due to habitat biases, [30]). Therefore, we used our own correction equations instead of others recently published (e.g., [32]). Observers were previously trained in order to minimize potential biases due to their different levels of expertise in carcass searches [25].

Table 2. Equations from [25] used in our study to correct numbers of dead birds found at the power line, in order to account for removal by scavengers or missed observations during carcass searches. Different equations are given for the four size categories specified in [25] (see Table 3 for their weights). We first corrected the number of carcasses found in the field by their size-dependent detectability (A). Second, we applied equation B for different carcass sizes where "days" is the number of days elapsed from the last visit. Third, we obtained a correction for every size category. Finally, we added C to A to obtain the mortality estimates for each category. The mortality estimate for a given power line was the sum of mortality estimates for the four carcass sizes.

Equation	
A_n (Detectability)	A_1 : Large= (no. carcasses found+1)*100/71.7 A_2 : Medium= (no. carcasses found+1)*100/55.8 A_3 : Small= (no. carcasses found+1)*100/32.1 A_4 : Very small= (no. carcasses found+1)*100/33.3
B_n (Periodicity and scavenging)	B_1 : Large = 0.744+28.063*log10(days) B_2 : Medium=-1.751+41.880*log10(days) B_3 : Small=-6.623+58.111*log10(days) B_4 : Very small=13.538+60.342*log10(days)
C_n (Correction)	$(A_n*B_n)/100$
Mortality estimate $_n$	$A_n + C_n$

In addition to testing the effectiveness of line marking as a means to reduce bird collision rate, we also evaluated two potential sources of variation in marking efficiency: power line type and spiral size. Whereas all transmission lines were equipped with large spirals (35 cm diameter and 1 m length, Figure 1a), either large or small spirals (10 cm of diameter and 24 cm m long, Figure 1b) were attached to distribution lines, with the same spiral size attached to all the spans of a given power line. We compared (i) the differences in marking efficiency in transmission *vs*. distribution lines when equipped with large spirals; and (ii) the efficiency of large *vs*. small spirals to reduce bird mortality in distribution lines.
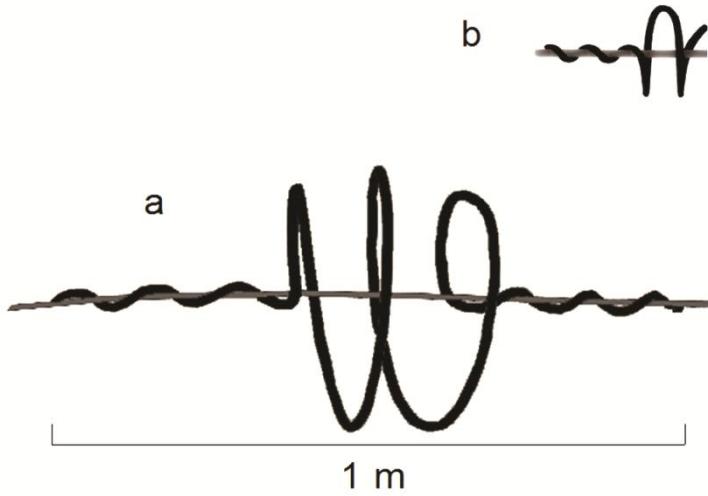
Figure 1. Spirals used in our experiments. Difference in size between large (a) and small (b) can be appreciated.

Unfortunately, we have no data on flight frequencies to estimate collision rates associated with our different designs, but in the study of marking effectiveness alone we used the corresponding *controls* to evaluate potential changes in bird mortality associated with changes in bird population densities. Furthermore, power lines of different categories were surveyed in the same study area, minimizing the effect of potential local differences in bird densities.

Statistical analyses

As a basic first analytical approach we tested whether there was a trend in the number of bird carcasses found after marking the line compared to before marking. This was done considering each power line as a sample unit, and comparing the number of decreases and increases in casualties recorded after marking (in the case of experimental lines), or in the second survey year compared to the first year (in the case of control lines). These comparisons were performed using the two-tailed sign test for small samples [33]. The same test was carried out using the total *estimated* number of dead birds, i.e. after correcting the number of casualties recorded during the field surveys [25]. To confirm the observed trends, we checked the differences in the accumulated numbers of *estimated* deaths

before-after marking (first-second year in the case of controls) and experimental lines-control lines by means of a chi-squared test.

As a second approach we used a Generalized Linear Mixed Model (GLMM) of various independent factors on the monthly estimated collision rate, after applying corrections proposed by [25] to the number of carcasses found to account for carcass losses due to removal by scavengers or to being overlooked by observers. For this analysis we considered one month as a time lapse long enough to allow the use of carcass search results in different months as statistically independent. We performed three GLMMs with Poisson error distributions and log link functions. The three analyses shared the same dependent variable, the *estimated* number of dead birds per month, and standardizing per kilometre of power line [30]. They also shared the random factor (power line). The models were fitted by maximizing the log-likelihood using the Laplacian approximation in R-Program 2.11.1 ([34]; *lmer* in *lme4* package). The three analyses were the following: (i) Marking effectiveness alone: We evaluated the effect of wire marking on bird mortality with two fixed factors, 'Marked *vs.* non-marked', with two levels, and 'First survey year *vs.* second survey year', also with two levels. This analysis includes both lines marked in the second year, but not in the first, and control lines. (ii) Power line type: We explored the effect of the power line type by including a factor with two levels (transmission and distribution) in the sample of power lines marked with large spirals. (iii) Spiral size: We studied the effect of spiral size through a factor with two levels (large and small) in the sample of distribution power lines.

In order to evaluate the importance of correcting for corpse losses, we performed a *sensitivity analysis* with a second group of GLMM tests where the dependent variable was the raw number of carcasses (i.e., those found in the field, without correction per losses) per km per month. All other parameters remained constant. This was only a methodological approach, as all the findings were based on the above-mentioned *estimated* mortality.

Finally, to study the specificity of the patterns found, we re-analyzed our data from a species-specific point of view. However, most of the species did not allow analyzing them with a GLMM procedure because they were not well

represented in all the power lines along the study area. We thus proceeded with Wilcoxon paired-sample tests for the three most common species: (i) doves (rock and domestic doves and wood pigeons, all together), (ii) great bustards and (iii) little bustards. We took into account the changes in mortality (first year *vs*. second year) for the whole power line and separating experimental and control lines. We made these species-specific calculations after correcting the number of casualties recorded during the field surveys, i.e., with *estimated* mortality.

RESULTS

We found 521 carcasses of 45 bird species, 19 of conservation concern (Table 3). Among experimental lines, most showed a decline in mortality after line marking compared to before line marking (11 lines with a decrease, 4 with an increase; *P*= 0.10, two-tailed sign test). The overall decrease in the number of carcasses recorded in the sample of 15 experimental lines was 88 birds (189 birds before marking, 101 birds after marking, 47% reduction in observed casualties). In control lines we did not observe a significant trend (10 lines with a decrease, 5 with an increase, 5 remained constant, *P* = 0.30, two-tailed sign test), with an overall reduction of 20%.

Table 3. Species found dead under power lines in the present study and their size following [25]: XS (<50g), S (50-150g), M (150-600g) and L (>600g). Figures are numbers of carcasses found during the whole study period (2001-2010). Note that statistical analyses were made both with raw data and after applying correction equations proposed by [25] to field data shown in this table. The conservation status is based on [43] criteria: 'SPEC 1': European species of global conservation concern; 'SPEC 2': Species having global populations concentrated in Europe and an unfavourable conservation status in Europe; 'SPEC 3': species having global populations not concentrated in Europe but an unfavourable conservation status in Europe; and, 'Non-SPEC': species having global populations not concentrated in Europe and a favourable conservation status in Europe.

Species	Size	Carcasses found	SPEC
Cattle Egret *Bubulcus ibis*	L	9	Non-SPEC
White Stork *Ciconia ciconia*	L	24	SPEC 2
Mallard Anas platyrhynchos	L	4	Non-SPEC
Shoveler Duck A. clypeata	L	1	Non-SPEC
Black Kite Milvus migrans	L	2	SPEC 3
Cinereous Vulture Aegypius monachus	L	2	SPEC 1
Marsh Harrier Circus aeruginosus	L	1	Non-SPEC
Sparrowhawk Accipiter nisus	M	1	Non-SPEC
Common Buzzard Buteo buteo	L	1	Non-SPEC
Common Kestrel Falco tinnunculus	M	6	SPEC 3
Red-legged *Partridge Alectoris rufa*	M	10	SPEC 2
Common Quail *Coturnix coturnix*	S	3	SPEC 3
Common Moorhen *Gallinula chloropus*	M	2	Non-SPEC
Little Bustard Tetrax tetrax	L	57	SPEC 1
Great Bustard Otis tarda	L	73	SPEC 1
Stone Curlew *Burhinus oedicnemus*	L	12	SPEC 3
Lapwing Vanellus vanellus	M	19	Non-SPEC
Black-headed Gull Larus ridibundus	L	2	Non-SPEC
Pin-tailed Sandgrouse *Pterocles alchata*	M	6	SPEC 3
Rock/Domestic Dove *Columba livia*	M	130	Non-SPEC
Wood Pigeon *C. palumbus*	M	49	Non-SPEC
Common Swift *Apus apus*	S	1	Non-SPEC
European Roller *Coracias garrulus*	S	4	SPEC 2
Crested Lark *Galerida cristata*	XS	1	SPEC 3
Skylark *Alauda arvensis*	S	14	SPEC 3
Barn Swallow *Hirundo rustica*	XS	1	SPEC 3
Meadow Pipit *Anthus pratensis*	XS	7	Non-SPEC
Robin *Erithacus rubecula*	XS	1	Non-SPEC
Northern Weather *Oenanthe oenanthe*	XS	1	SPEC 3
Blackbird Turdus merula	S	1	Non-SPEC
Reed Warbler *Acrocephalus scirpaceus*	XS	1	Non-SPEC
Melodious Warbler Hippolais polyglotta	XS	1	Non-SPEC
Subalpine Warbler *Sylvia cantillans*	XS	3	Non-SPEC

Species	Size	Carcasses found	SPEC
Orphean Warbler *S. hortensis*	XS	1	SPEC 3
Blackcap *S. atricapilla*	XS	2	Non-SPEC
Common C*hiffchaff Phylloscopus collybita*	XS	4	Non-SPEC
Willow Warbler *P. trochilus*	XS	3	Non-SPEC
Magpie *Pica pica*	M	28	Non-SPEC
Jackdaw *Corvus monedula*	M	1	Non-SPEC
European *Starling Sturnus vulgaris*	S	1	SPEC 3
Spotless Starling *S. unicolor*	S	8	Non-SPEC
House Sparrow *Passer domesticus*	XS	3	SPEC 3
European Serin *Serinus serinus*	XS	1	Non-SPEC
Linnet Carduelis cannabina	XS	3	SPEC 2
Corn Bunting *Emberiza calandra*	XS	7	Non-SPEC
Undetermined medium-sized bird	M	3	---
Undetermined passerine	XS	6	---

The 521 dead birds found represent 14,282 estimated bird collisions, an average 8.2 collisions per month and km, after accounting for carcass removal by scavengers and missed observations during surveys. Significantly more experimental lines showed a decrease in the number of estimated casualties after line marking compared to before line marking (12 lines with a decrease, 3 with an increase; P= 0.04, two-tailed sign test). The overall difference in the sample of 15 lines was 316 birds (3,300 estimated birds before marking, 2,984 birds after marking, 9.6% reduction in estimated mortality). The control sample did not show significant before-after differences (10 lines with a decrease, 10 with an increase, P= 1.0, two-tailed sign test; total estimated casualties: 4,067 before and 3,931 after marking, 3.3% reduction). A chi-squared test with the former data (3,300, 2,984, 4,067 and 3,931) confirmed the difference between experimental and control samples in the reduction of estimated casualties (χ^2 = 3.90, P = 0.048).

In the GLMM considering all monthly surveys, the number of estimated collisions per kilometre was significantly reduced in experimental power lines after marking, while it remained similar in controls (Table 4i.a; Figure 2). This model explained 96.4% of the deviance. The effectiveness of large spirals was similar in transmission and distribution power lines (Table 4ii.a; Figure 3). The model explained 99.6% of the deviance. Spirals of different sizes had similar marking effectiveness when attached to distribution lines (Table 4iii.a; Figure 4), with 98.8% of the deviance explained by the model. The comparisons with uncorrected raw data (Table 4i.b, ii.b and iii.b) showed different statistical differences (e.g., in 'marked *vs*. non-marked'), highlighting the importance of correcting field data.
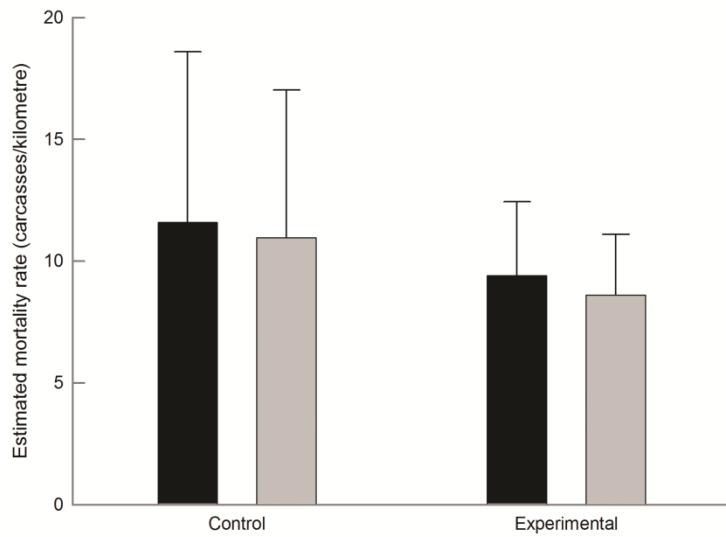
Figure 2. Number of estimated carcasses per kilometre (mean ± SE) before (black) and after (grey bars) marking in control (left) and experimentally marked (right) power lines. Sample sizes were 219 and 165 in each period for control and experimental power lines, respectively.
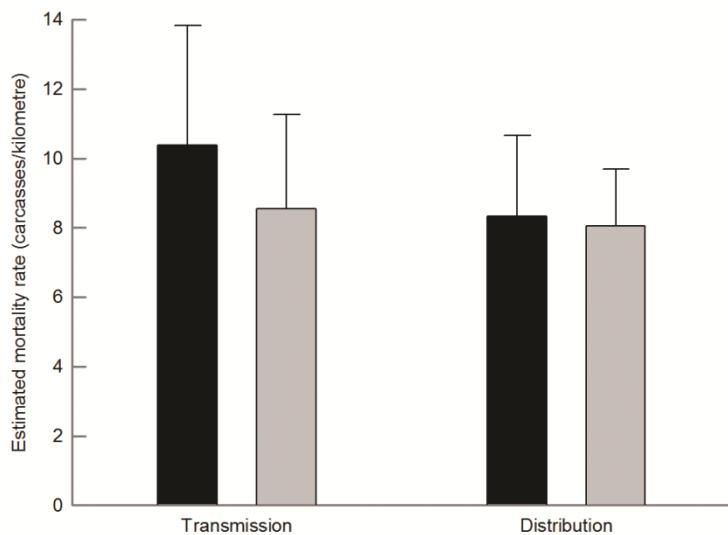
Figure 3. Number of estimated carcasses per kilometre (mean ± SE) before (black) and after (grey bars) marking in transmission (left) and distribution (right) power lines. Sample sizes were 77 and 44 in each period for transmission and distribution power lines, respectively.

Table 4. Parameter estimates from the Generalized Linear Mixed Model for marking effectiveness alone model (i), power line type model (ii) and spiral size model (iii). We show GLMM with (a) and without (b) corrections for carcass losses due to researcher overlooking and removing by scavengers. Estimate, standard error (SE), statistic value (z) and statistical significance (*P*) are provided.

	Estimate	SE	z	*P*
(i.a) Marking effectiveness alone (n=770) (with corrections)				
Intercept	2.34	0.09	27.31	<0.0001
Marked *vs*. non-marked	-0.08	0.04	-2.13	0.03
First survey year *vs*. second survey year	-0.04	0.03	1.57	0.12
(i.b) Marking effectiveness alone (n=770) (without corrections)				
Intercept	-1.20	0.20	-6.35	<0.0001
Marked *vs*. non-marked	-0.30	0.16	-1.90	0.06
First survey year *vs*. second survey year	0.47	0.14	3.46	<0.0001
(ii.a) Power line type (n=242) (with corrections)				
Intercept	2.10	0.11	18.49	<0.0001
Power line type	0.11	0.14	0.78	0.44

(ii.b) Power line type (n=242)
(without corrections)

	Estimate	SE	z	P
Intercept	-1.71	0.32	-5.42	<0.0001
Power line type	0.75	0.38	1.99	0.05

(iii.a) Spiral size (n=176)
(with corrections)

	Estimate	SE	z	P
Intercept	2.10	0.08	25.12	<0.0001
Spiral size	0.10	0.12	0.88	0.38

(iii.b) Spiral size (n=176)
(without corrections)

|	Estimate	SE	z	P
Intercept	-1.75	0.36	-4.92	<0.0001
Spiral size	0.65	0.49	1.32	0.19
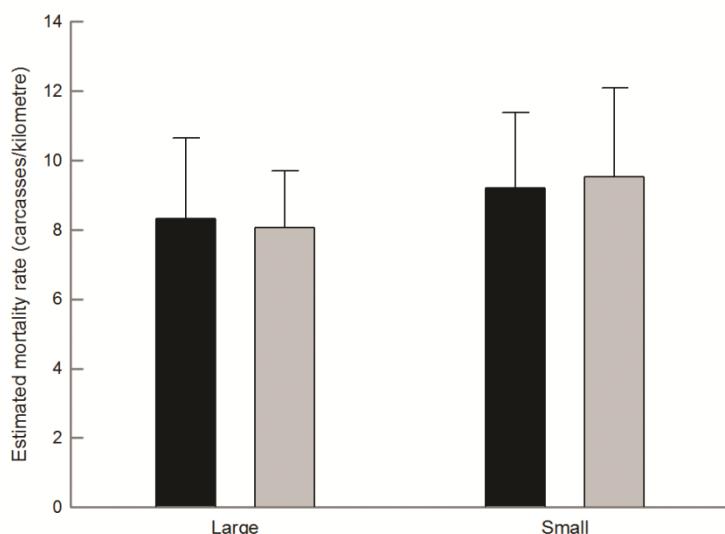
Figure 4. Number of estimated carcasses per kilometre (mean ± SE) before (black) and after (grey bars) marking in distribution power lines marked with large (left) and small (right) spirals. See Figure 1 for more details. Sample sizes were 44 in all cases.

Regarding species-specific patterns, doves did not show significant differences in the six treatments, regarding marking effectiveness alone (Wilcoxon paired-sample test, marked *vs.* non-marked, $Z = 0.87$, $P = 0.39$; first survey year *vs.* second survey year, $Z = 0.00$, $P = 1.00$), power line type (transmission lines, $Z = 0.41$, $P = 0.68$; distribution lines, $Z = 0.41$, $P = 0.68$) or spiral size (large spirals, $Z = -0.32$, $P = 0.75$; small spirals, $Z = -0.50$, $P = 0.62$).

In contrast, great bustard mortality was reduced only after marking of transmission lines (transmission lines, Z = 2.04, P =0.04; distribution lines, Z = 0.00, P =1.00) or only when marking with large spirals (large spirals, Z = 2.00, P =0.046; small spirals, Z = -0.71, P =0.48), being not significant regarding marking effectiveness alone (marked *vs.* non-marked, Z = 1.81, P =0.07; first survey year *vs.* second survey year, Z = 0.00, P =1.00).

In the little bustard, wire marking reduced mortality (Z = 2.47, P =0.01), whereas statistical differences were not found for controls (Z = 0.50, P =0.62) or for power line type (transmission lines, Z = 1.79, P =0.07; distribution lines, Z = 1.15, P =0.25) or spiral size (large spirals, Z = 1.22, P =0.22; small spirals, Z = 0.00, P =1.00).

DISCUSSION

Our results show a slight (overall, 9.6%, after correcting for carcass removal by scavengers and missed observations), but significant reduction in bird mortality after flight diverters were attached to power lines. Regardless of statistical significance, a slight mortality reduction may be very biologically relevant in areas, species or populations of high conservation concern. It is important to note that overall mortality reduction values were not the same if calculated using raw numbers of dead birds found, i.e. before correcting for carcass removal by scavengers and missed observations. This is because correction factors differ between species [25]. Thus, uncorrected mortality values would lead to incorrect conclusions, and special care should be taken when dealing with certain birds of conservation concern. Neither the type of line (transmission *vs.* distribution) marked with large spirals, nor the size of spirals in distribution lines influenced the magnitude of mortality reduction when we assessed overall mortality in all species together. However, great bustard mortality showed reductions when lines were marked with large spirals, and also considering only transmission lines.

The effectiveness of wire marking in reducing bird mortality through collision has been recently reviewed by Barrientos *et al.* [19]. However, in that study, different markers were combined since available sample sizes did not allow

inclusion of marker type as a factor in the analysis. Thus, despite spirals of different sizes and colours being the most frequently employed bird flight diverters, half of the studies included in Barrientos *et al*. [19] referred to other device types (see Appendix in [19]). The present study suggests that the mortality reduction found in that review was not due to the inclusion of other markers, and that the most widely used spirals are effective. The present study also overcomes a common problem detected in Barrientos *et al*. [19], namely that sample sizes are generally small. Here we based our conclusions on a large sample including two-year monthly surveys at 15 experimental and 20 control power lines, covering 72.5 km. Moreover, these lines were distributed over a relatively large geographical area, encompassing most farmland areas used by steppe birds in our study region. This overall low (9.6%) reduction could be greater in some places (e.g., migration corridors, power lines close to resting sites, etc), or could represent a valuable reduction for endangered species with high collision risk. Thus, a detailed evaluation of mortality due to collision should be carried out before deciding where to attach spirals as a bird protection measure in relatively large conservation areas.

Some of the species found dead in our study are among those suggested in previous studies to be the most likely to collide with power lines [2,8], namely those with low maneuverability such as bustards, storks or waterfowl. These species usually fly higher than, for instance, many passerines, and thus most of their collisions are expected to be with transmission lines. Indeed, if we consider the data from the first year only, i.e. before attaching spirals, transmission lines in our study accumulated 71% (n=42) of all great bustards found dead in all lines, 50% (n=50) of all little bustards *Tetrax tetrax*, 83% (n=12) of all white storks *Ciconia ciconia* and 100% (n=3) of all ducks *Anas* spp., despite the fact that transmission lines represented only 36% of the total length of power lines surveyed. In their study with whooping cranes, Morkill & Anderson [22] found that birds reacted more than expected to transmission lines and less to distribution lines. However, we did not find a significant difference in mortality reduction in marked transmission lines compared to marked distribution lines when we considered all species together. When looking at species-specific patterns, only the great bustard showed a slight mortality reduction in marked transmission lines.

Although some studies found that species suffering high collision mortality may show a tendency to avoid areas with transmission lines (e.g. little bustard, [35]), collision with transmission lines is still one of the most important sources of mortality in these species [35, 36]. Thus, as suggested in Barrientos *et al*. [19], it is possible that at least some of these particularly sensitive species do not properly respond to conventional marking methods (see below).

Although one would expect that large flight diverters are more effective than small diverters in increasing the visibility of marked wires, other authors that have used spirals of different sizes [23,24] did not statistically test for differences among them. Our study explores this possibility for the first time. Considering all species together, our results suggest that the decrease in collision rate is independent of spiral size, and thus it seems reasonable to conclude that the main advantage of marking is already achieved with small spirals, with larger spirals being unnecessary. This could imply interesting applied findings. For example, small diverters do not apply excessive weight to the wire. Large devices can constitute a problem for this reason especially in high winds, contributing to the downing of power lines, especially if devices are frozen [14,22]. However, a flagship species like the great bustard showed mortality reduction with larger spirals, suggesting that, at least for this species, large spirals work better.

Despite our study being, to our knowledge, the largest published field experiment, and ca. 310,000 € were spent to mark 33.7 kilometres of power lines in our study area, few conclusions can be drawn beyond the general effectiveness of bird flight diverters in reducing collision mortality. We found differences in effectiveness when we compared markers in transmission versus distribution lines, or when we compared spirals of different sizes in distribution lines only with one species (although we could carry out species-specific analyses only with three species). However, it is worth noting that even after marking, bird collisions in our study area were still high, especially for some endangered species usually showing high collision risks (e.g. great and little bustards). Several non-mutually exclusive explanations could account for this. First, it is possible that the generally low probability of collision (0.21-0.05 birds per 1,000 crossings; [19]) makes it very difficult to find differences even with well-designed experiments. If this is the case,

huge experimental designs would be necessary to find larger differences and extract stronger conclusions. Second, it has been argued that bad weather or light conditions can increase bird collisions, especially if birds have problems with flight control [14,37]. For most birds, sustained slow flight is costly or aerodynamically impossible [38, 39], and hence reducing speed is an unlikely mechanism to increase safety under bad weather or light conditions. Third, collisions frequently occur even under low wind and good visibility conditions [40]. Recent studies [9,10] suggest that some species, which undergo high collision rates (e.g. bustards and storks) have narrow fields of view in the frontal plane, hindering their ability to see the way ahead. Fourth, Martin [10] suggests that birds flying in open airspace above vegetation could relax –by means of either behavioural or evolutionary adaptations- the monitoring of this airspace since it is a highly predictable environment, usually clear of hazards. In other words, birds of some species could simply not look ahead during flight. Indeed, frontal vision in birds is not a high-resolution vision [10]. Instead, the best resolution occurs in the lateral vision, which most birds employ to detect conspecifics (very important in social species like bustards or storks) and predators, or in identify foraging opportunities. All of these may be more important for a bird than simply looking ahead during flight into open airspace [10]. Fifth, anecdotal events can have potentially important effects on collisions. For instance, Sastre *et al.* [41] suggest that human-related disturbances causing flight response can increase the probability of collision of great bustards with power lines. Sixth, regarding the effectiveness evaluation of different devices, it is also plausible that misguided approaches have been used to date. For instance, whereas bird flight diverters are usually coloured with a single colour bright to the human eye [19], a recent review [10] recommends the use of black-and-white diverters, which reflect highly or absorb strongly across the full spectrum of ambient light. Thus, it is possible that the few valuable studies carried out to date that compared the effectiveness of different colours for a certain bird flight diverter [42] actually compared colours too close in the spectrum to identify differences in their effectiveness. Since it is recognized that the colour vision of birds extends into the ultraviolet range, thus broadening, compared with humans, the range of stimuli to which the avian eye can respond [10], the use of ultraviolet-devices should be investigated.

In summary of the above-mentioned explanations, and given that is seems clear that no single type of marker will be equally effective for all bird species, we acknowledge that the importance of type and size of bird flight diverters is not yet clear and should be confirmed in future studies. Our study does not pretend to be comprehensive in this respect, and regarding the different susceptibilities of different bird species or groups to collision [see 2,8], and particularly the mortality reductions obtained for specific models of flight diverters should be further investigated. In this sense, we encourage researchers to explore the effectiveness of non-visual diverters. Finally, we highly recommend the identification of mortality hot-spots based on the number of individuals killed and the vulnerability of the species involved [e.g. 44]. Taking into account the economic cost of marking, it is likely more useful to attach flight diverters to these hot-spots rather than to do it to whole sections of power line.

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CAPÍTULO 3

Este capítulo reproduce íntegramente el siguiente artículo:

Ponce, C., Bravo, C., García de León, D., Magaña, M. & Alonso, J.C. 2011. Effects of organic farming on plant and arthropod communities: a case study in Mediterranean dryland cereal. *Agriculture, Ecosystems & Environment* 141: 193–201.

CAPÍTULO 3

Effects of organic farming on plant and arthropod communities: a case study in Mediterranean dryland cereal

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ABSTRACT

Organic farming is considered an important way to preserve biodiversity in agricultural landscapes. However, more work is still necessary to enable a full appraisal of the potential benefits of this way of farming, since studies differ in the evaluation of its effectiveness. Studies are particularly scarce in the Mediterranean region, where different climatic and ecological conditions prevent simple extrapolations from work carried out at northern latitudes. In the present study, an analysis of weed and arthropod communities was conducted in 28 pairs of organic and conventional fields in a dry cereal farmland in central Spain. Plants were identified to the species level, and arthropods to the family level. Pitfalls and sweep nets were used to sample respectively, ground-dwelling and plant-visiting arthropods. Abundance (total numbers of individuals), richness (total numbers of plant species or arthropod families), diversity (Shannon-Wiener index) and biomass (milligrams per pitfall/sweep-net) were calculated for each field and compared between organic and conventional fields using Generalized Linear Mixed Models (GLMMs). To explore the effect of predictor variables on weed richness and arthropod biomass, GLMMs were used. Organic fields showed higher abundance of weeds and arthropods (respectively, 3.01 and 1.43 times), higher weed richness and diversity (respectively, 2.76 and 2.33 times), and a 24% reduction in cereal plants. Arthropod diversity was lower in organic fields due to the presence of three dominant groups: Collembola, Chloropidae (Diptera), and Aphididae (Hemiptera). Weed richness increased as cereal cover decreased in organic fields. Total arthropod biomass was slightly higher in organic fields, and was affected by weed abundance and diversity. The differences between organic and conventional fields found in this study were higher than those reported for northern latitudes. This could be explained by the richer weed flora in the Mediterranean region, and a higher weed seed availability favoured by the two-year rotation system typical of Iberian dry cereal farmland. We conclude that organic farming may contribute to preserve biodiversity in dryland cereal agroecosystems in the Mediterranean region.

Keywords

INTRODUCTION

A wealth of evidence points to agricultural intensification as the main cause of biodiversity loss in farmland ecosystems (Donald *et al.*, 2006; Foley *et al.*, 2005; Millennium Ecosystem Assessment, 2005; Wilson *et al.*, 2009, 2010). This negative impact of modern agriculture on many plant and animal taxa will probably raise in the future, due to increasing demands in agricultural production. This is at present an issue of major concern worldwide (Clough *et al.*, 2007a; Fuller *et al.*, 2005; Hole *et al.*, 2005), and there is a growing consensus that further increases in agricultural production must avoid further adverse environmental impacts (Firbank, 2009; Royal Society, 2009). One of the ways to reverse this negative trend would be to use organic farming methods (Geiger *et al.*, 2010). Agri-environment schemes including organic farming and other environmentally friendly practices are today considered the most important instruments to counteract the negative effects of modern agriculture (EEA, 2004). However, published studies differ in their evaluation of the effectiveness of these measures, which makes it difficult to assess their benefits (Bengtsson *et al.*, 2005; Frampton and Dorne, 2007; Kleijn *et al.*, 2006).

In a comprehensive review of comparative studies of organic and conventional farming systems, Hole *et al.* (2005) found inconsistencies between and within studies which suggested that the benefits to biodiversity of organic farming may vary according to factors such as location, climate, crop-type and species. They concluded that further studies are still needed in order to understand the impacts of organic farming, before a full appraisal of its potential role in biodiversity conservation in agroecosystems can be made. For example, many recent studies have attempted to evaluate the effectiveness of organic farming using birds, plants or invertebrates as study subjects (Beecher *et al.*, 2002; Bengtsson *et al.*, 2005; Diekötter *et al.*, 2010; Chamberlain *et al.*, 2010; Clough *et al.*, 2005; Clough *et al.*, 2007a; Fuller *et al.*, 2005; Gabriel *et al.*, 2006; Gabriel *et al.*, 2010; Gibson *et al.*, 2007; Piha *et al.*, 2007; Roschewitz *et al.*, 2005; Schmidt *et al.*, 2005; Weibull *et al.*, 2003). However, most of these studies have been carried out at mid- or high latitudes of the northern hemisphere, and very few in the Mediterranean Region, where climatic conditions are quite different (e.g., lower

rainfall, higher temperatures, lower soil organic content, and considerable variation in the amount of water available for different springs; Costa *et al.*, 2004; INE, 2009; Walter, 1994), making it difficult to extrapolate the conclusions from northern latitudes (Hole *et al.*, 2005).

Two recent studies address the effect of organic farming on arthropods in the Mediterranean Region, but not in dryland cereal fields (Cotes *et al.*, 2010; Hadjicharalampous *et al.*, 2002). In this Region, only three studies used vascular plants as study subjects. In José-María *et al.´*s (2010) study, management was the main factor explaining differences among field centres, while Romero *et al.* (2008) found that organic farming increased weed cover, and species richness and diversity. Another study carried out in four organically managed fields (Caballero-López *et al.*, 2010) showed that plants are highly dependent on farming system, and the arthropod community is conditioned by those plants, which led the authors to conclude that interactions are also important in order to assess the importance of management in cereal crops. Finally, in their recent review, Hole *et al.* (2005) stated the need of further studies particularly in the Mediterranean region.

In the present study we evaluated the effects of organic farming on biodiversity in a dry cereal farmland in central Spain. The aim was to determine whether there were any differences in the weed and arthropod communities between fields that had been farmed without using synthetic fertilizers and pesticides (organic system), and fields where these chemicals were used (conventional system). Therefore, unlike most previous studies that concentrated on single plant or invertebrate groups, we quantified the effect of the agro-chemical treatment on the abundance (total numbers of individuals), richness (total numbers of species or families), and diversity (Shannon-Wiener index) of all identifiable vascular plants and arthropods found, as well as on the cover of grown cereal and weeds, and on arthropod biomass. Besides, we characterized the factors affecting both weed richness and arthropod biomass, since these are some of the most studied variables in organic farming studies.

METHODS

Study area, field selection and farming practices

The study was conducted in 2008 in a Special Protection Area for birds (SPA 139, 'Estepas Cerealistas de los ríos Jarama y Henares') about 25 km north of Madrid (40º42'N, 3º29'E; 682 m.a.s.l.), in central Spain. The terrain is flat to slightly undulated, and it is primarily dedicated to dryland cereal cultivation (wheat *Triticum aestivum* (L.), barley *Hordeum vulgare* (L.), and smaller amounts of common oat *Avena sativa* (L.), together more than 95% of the surface), with minor fields of legumes (*Vicia spp. and Medicago sativa* (L.)), olive groves *Olea europaea* (L.) and grapevines *Vitis vinifera* (L.). The brown and acid soil present in the study area and the weather conditions favor a natural vegetation composed by evergreen oak trees (*Quercus ilex* (L.); and their degraded states –*Retama sp.* and *Thymus sp.* scrubland-), which instead of forming dense woods have been cleared up to open-wooded area called 'dehesas' used for wood extraction and livestock grazing. Scattered groups of white poplars (*Populus alba* (L.)) are also found in the SPA, although as in the case of oaks, always more than 1 km away from our sampling fields, and thus probably having no influence on them. Most cereal is grown in a traditional two-year rotation system, and harvested during late June-early July. The climate is Mediterranean, with an annual precipitation (mean ± S.D.) of 442.5 ± 125.5 mm and a mean annual temperature of 14.4 ºC (maximum and minimum temperatures, respectively, 42.2 ºC and -14.8 ºC). During the study year, the mean annual precipitation was 484.9 mm and the mean monthly temperature, 14.3 ºC (maximum and minimum temperatures, respectively, 39.3 ºC and -6 ºC). The mean temperature during May is 15.6 ± 1.6 ºC, and the mean rainfall, 55.1 ± 41.2 mm. In May 2008 these values were, respectively, 15.5 ºC and 64.7 mm, so we can consider our study year as normal. The study area is a SPA for birds because it holds significant populations of globally threatened steppe birds. Therefore, an agri-environmental scheme is running in this area since 2001, as part of the compensatory measures for the construction of a highway crossing its southern margin. Organic farming was one of the conservation actions implemented in a sector of the SPA.

Twenty-eight pairs of fields were randomly selected, where one field of each pair was cultivated without synthetic fertilizers and pesticides (organic system), and the other field with such products (conventional system) (see e.g. Clough *et al.*, 2007b; Pfiffner and Niggli, 1996; Shah *et al.*, 2003). All sampled cereal fields (always dedicated to cereal cultivation) were preceded by a fallow year before the study was carried out, so the initial conditions were the same for all of them and the only difference was that one field of the pair was cultivated organically during the year when our study was conducted. Fields of the same pair were separated by <100 m and shared the major physiographic characteristics (slope, orientation, approximate size, soil type) and farm history. The mean field size was 1.9 ± 0.9 ha, similar to that of a previous study in northern Spain (José-María *et al.*, 2010). Since the maximum distance between fields in our sample was 11 km, we considered that the environmental conditions were the same for all fields.

Farmers were asked to fill out a questionnaire to characterize their usual farming practices, which are compared to those allowed in organic fields (Table 1). Both organic and conventional fields were sown (wheat or barley) between the second week of October and the first week of November 2007, after initial ploughing for soil preparation. Conventional fields were later treated with chemical fertilizers (Table 1) and broad-leaf herbicides, while organic fields did not receive such treatments. The density of seeds (wheat or barley) was the similar in both, organic and conventional fields ($T = 1.80$, $P = 0.12$, Table 1).

Table 1. Main characteristics of the farming system used in the 28 pairs of fields.

	Organic fields	Conventional fields
Sowing density (wheat or barley)	188 ± 16 kg ha^{-1}	197 ± 19 kg ha^{-1}
Fertilization	No	NPK: 350 ± 72 kg ha^{-1}, October CAN (27%): 168 ± 26 kg ha^{-1}, February
Weed control	Weed ploughing	Weed ploughing Clorsulfuron (7%): 2-2.5 g ha^{-1}. April, May and July Clortoluron (50%): 3-4 l ha^{-1} Gardel: 0.2 l ha^{-1} Foramsulfuron: 10 g ha^{-1}. April, May and July Primafuron: 20 g ha^{-1}
Seed origin	Organic	Industrially selected and chemically treated
Ploughing (mouldboard plus weed ploughing)	1-2 times/year	2-4 times/year

Plant and arthropod sampling

Plant sampling was carried out during the third week of May 2008. A 25 x 25 cm metal square was thrown randomly 20 times in each field, avoiding the edges and their proximities. Each plant was identified to the species level, the number of individual plants of each species was counted and the corresponding cover for each species estimated as a percent of the square surface. In the case of cereal, the total number of plants was used as an indirect measure of cereal production, since no information about crop could be obtained. To check if plant sampling effort was sufficient, species accumulation curves were generated using the program EstimateS version 8.2 (Colwell, 2009) and fitted by Clench equation (Jiménez-Valverde $et\ al.$, 2003; Moreno and Halffter, 2001). The Clench equation was defined as $S_n = AxN/(1+BxN)$, where S_n is the number of species observed in each given sample level, A is the increase rate of new species at the beginning of sampling and B is the parameter related to shape of the curve. The asymptote of curve -total number of species predicted- is calculated as A/B.

We used two different methods to sample arthropods, pitfall traps and sweep nets. Pitfall traps are the most appropriate method to capture terrestrial

and soil arthropods (e.g., Clere and Bretagnolle, 2001; Hadjicharalampous *et al*., 2002; Schmidt *et al*., 2006), while sweep nets are commonly used for taxa such as Heteroptera that are living well above the ground on the plant canopy, or those that spend much time flying (Frampton and Dorne, 2007). The combination of both methods provides the best possible information about the arthropod fauna (Fauvel, 1999). Within each field, three pitfall traps were placed during the third week of May close to the field center at 10-m intervals. Each trap consisted of a plastic cup (9 cm internal diameter, 14 cm length) sunk into the soil with the aid of a metal cylinder, and filled with 250 ml of 70% ethanol as a preservative solution (Shah *et al*., 2003). Traps were protected from rainfall and excessive evaporation by plastic dishes suspended on thin sticks at 10 cm over the soil surface. Collections of arthropods were made for 7 days (± 2 hours). Collected arthropods were stored in 70% ethanol after the sampling period. Five days after collecting the pitfalls, we conducted three sweep-netting transects on each field. Fields of the same pair were sampled one after each other, between 6:00 h and 10:00 h GMT, avoiding inappropriate weather conditions such as wind and temperature below 18°C or above 25°C, when arthropods might be inactive (Weibull and Östman, 2003). Each transect consisted of ten movements of the sweep net, from right to left side and vice versa and approximately 2 m wide. Before starting these samplings, all observers spent one day standardizing these sweep net movements to prevent sampling biases due to differences in width, depth and speed, and the same observer always sampled both fields of a pair. The arthropods captured were fixed in 70% ethanol.

Laboratory procedures and statistical analyses

Plants were identified to the species level and arthropod to the family level, which is useful for all indexes used in this study (abundance, richness and Shannon-Wiener diversity index; Biaggini *et al*., 2007; Frampton and Dorne, 2007), as well as for biomass calculations (Hódar, 1996). To estimate arthropod biomass we first measured with a digital caliper (0.01 mm precision) the maximum body length of all adult arthropods captured excluding appendices (wings, antennae, ovipositors or legs), and calculated the average body size for each taxonomic group. To estimate the mean biomass of each group, we used the equations given by Hódar

(1996), which relate weight to body length in several arthropod groups of the Mediterranean region (general equation: $Y = a^{b1}(x)^{b2}$, where Y is the biomass, x the length, and a,b1 and b2, specific coefficients for each taxonomic group). Consequently we calculated the biomass of each group in each sampled field (see also Clere and Bretagnolle, 2001; Jiguet et al., 2000).

We compared each index (richness, abundance, diversity and biomass) by means of Generalized Linear Mixed Models (GLMMs) with the field pair as random factor (to control for spatial non-independence in the data; Littel et al., 2006) using the lme4 package (Bates and Maechler, 2010) of R-Program 2.11.1 (R Development Core Team, 2010).

The differences in frequency distribution of the most abundant weed species between organic and conventional management were analyzed using Chi-squared test. Richness, abundance and diversity (plus biomass for arthropods) were calculated independently for plants and for arthropods. Later, these indices were calculated separately for cereal plants, weeds, and all plants (Kleijn et al., 2006; Lundkvist et al., 2008; Sunderland and Samu, 2000). Finally, we calculated these indices again and repeated the GLMMs for the most abundant arthropod orders. To check for dominant groups, we used the index proposed by Berger and Parker (1970). This index accounts for the dominance of the most abundant groups (the higher the value, the more dominant group), considering all species in the assemblage (Caruso et al., 2007). We repeated the diversity calculations excluding dominant arthropod groups.

Since arthropod biomass is expected to be related to vegetation variables (e.g., Clough et al., 2007b), we performed simple correlations analysis to discard, if necessary, some highly correlated variables. Next, we performed Generalized Linear Mixed Models (GLMMs), using field pair as random parameter, with Poisson error distribution and log link function. Biomass was the dependent variable and we included management type (organic or conventional), weed abundance, weed richness, weed diversity and weed cover as plausible independent variables. We performed another GLMM (with field pair as random effect) where weed richness was the dependent variable, and cereal cover and management, plus their interaction, the explanatory variables.

To determine the best predictive models, Akaike's information criterion (ΔAICc <2) was used. We used AICc because the ratio between the number of observations and estimator variables was under 40 (Barrientos and Bolonio 2009; Burnham and Anderson 2002). To look for differences among models with ΔAICc < 2, an ANOVA test was performed. The models were fitted by maximizing the log-likelihood using the Laplacian approximation because this is the most suitable for small sample sizes (Moya-Laraño and Wise 2007).

RESULTS

Plants

A total of 4940 plants belonging to 51 weed species were recorded, (Appendix A). The frequency distribution of these species differed between organic and conventional fields (χ^2 = 8467.2, P < 0.001; Table 2). Only four weeds were found in organic fields in lower numbers than in conventional fields (*Lolium rigidum* (Gaudin), *Avena sterilis* (L.), *Polygonum aviculare* (L.), and *Filago lutescens* (Jord.); Table 2). According to the Clench equation, we sampled 80% and 89.8% of the total number of predicted species, respectively in organic and conventional fields. The most abundant family was *Gramineae*, with 71.1% of total weeds (respectively, 60.9% and 88.1% in organic and conventional fields). Next were *Compositae*, with 10.4% (respectively, 14.4% and 3.8% in organic and conventional fields) and *Leguminosae*, with 4.3% (respectively, 7.3% and 0.2% in organic and conventional fields). Of 51 weed species identified, 48 were found in organic fields and 28 in conventional fields.

Table 2. Most common weed species ordered by the frequency with which they were recorded in organically managed and conventional fields.

Species	Organic fields	Conventional fields
Lolium rigidum (Gaudin)	50.3	80.2
Galium tricornutum (Dandy)	7.1	1.8
Bromus diandrus (Roth)	6.6	4.2
Anacyclus clavatus (Desf.)	4.2	0.7
Conyza canadensis (L.)	4.0	0.6
Raphanus raphanistrum (L.)	3.8	0.6
Avena sterilis (L.)	3.0	3.3
Vicia sativa (L.)	2.8	0.0
Polygonum aviculare (L.)	2.6	2.9
Filago pyramidata (L.)	1.7	0.1
Trifolium angustifolium (L.)	1.7	0.0
Filago lutescens (Jord.)	1.1	1.2
Vicia spp.	0.9	0.1
Picnomon acarna (L.)	0.9	0.8
Lactuca serriola (L.)	0.8	0.2
Ornithopus compressus (L.)	0.7	0.1
Linaria viscosa (L.)	0.7	0.0
Spergula arvensis (L.)	0.7	0.0
Euphorbia serrata (L.)	0.6	0.0
Vicia ervilia (L.)	0.6	0.0
Others	4.9	3.2

Values are percentages of each species found in both field types.

GLMMs showed that weed richness, weed diversity, weed abundance and weed cover were significantly higher in organic than conventional fields (Table 3), whereas cereal plants grew in higher numbers in conventional fields (Table 3). Total plant abundance (cereal plus weeds) was higher in conventional fields, and cereal cover and total cover did not differ between organic and conventional fields (Table 3). Overall, there was a negative relationship between cereal cover and weed richness, although this relationship was only significant for organically managed fields (Fig. 1). The GLMM showed that weed richness was influenced by the management type, cereal cover and their interaction (Appendix B, Table 4).

The first two models are equally valid (ANOVA test not significant), but the first including the interaction and had a lower AICc.

Table 3. Differences between conventional and organic fields in abundance, cover, richness, and diversity of plants.

		Organic fields	Conventional fields	*Z*	*P*	*LL*
	Cereal	475.2 ± 242.6	623.6 ± 141.2	-23.6	<0.001	-953.2
Abundance	Weeds	132.5 ± 153.8	43.9 ± 75.5	35.38	<0.001	-1214
	Both	607.7 ± 241.9	667.5 ± 125.4	-7.69	<0.001	-785
	Cereal	23.5 ± 14.2	35.7 ± 12.2	-0.39	0.261	-110.8
Cover	Weeds	9.9 ± 9.8	1.9 ± 2.8	12.59	<0.001	-73.6
	Both	33.4 ± 14.1	37.6 ± 11.2	-0.89	0.143	-104.5
Richness	Weeds	9.4 ± 4.0	3.4 ± 1.8	8.63	<0.001	-36.60
Diversity	Weeds	1.4 ± 0.5	0.6 ± 0.5	3.05	0.002	-13.25

Abundance measured as individuals per 1.25 m^2, or twenty 25x25cm sampling units, cover as %, richness as species per 1.25 m^2, and diversity through Shannon-Wiener diversity index. Mean values ± SD, statistic (*Z*, GLMM-test), significance of the differences (*P*) and log-likelihood (*LL*) are given.
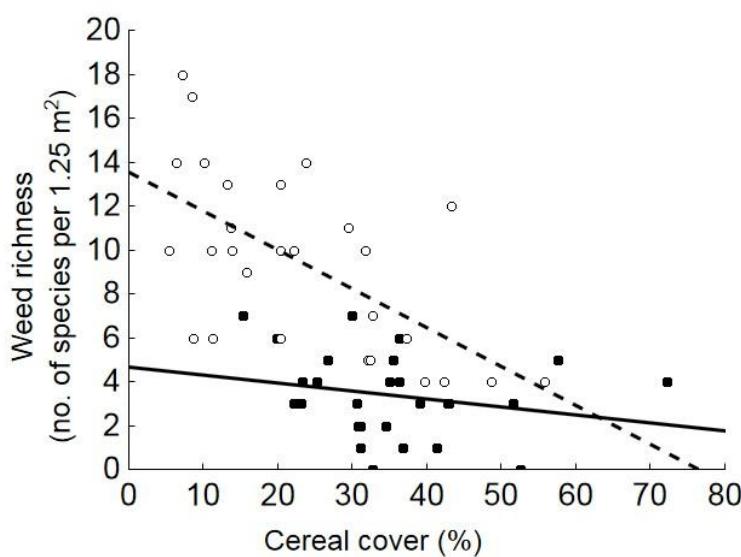
Figure 1. Relationship between cereal cover (%) and weed richness (no. of species per 1.25 m^2). The correlation was significant for organic fields (open circles; r = -0.62, *P* < 0.001), but not for conventional fields (black circles; r = -0.23, *P* = 0.23).

Table 4. Parameter estimates from the Generalized Linear Mixed Model with cereal cover (CC), management (MA), and the interaction between cereal cover and management (CC*MA) as factors affecting weed richness.

Parameter	Estimate	SE	P
CC	-0.009	0.052	0.452
MA	6.481	2.301	0.007
CC*MA	-1.873	0.325	0.031

Arthropods

A total of 82822 individuals belonging to 150 arthropod families and 21 orders were collected (Appendix C). Arthropods were more abundant in organic fields (50488 individuals, Table 5).

Table 5. Differences between conventional and organic fields in abundance, richness, diversity, and biomass of arthropods.

	Organic fields	Conventional fields	Z	P	LL
Abundance	1798.4 ± 1052	1253.9 ± 508	54.66	<0.001	-3861
Richness	45 ± 4.9	42.3 ± 4.29	6.98	0.002	-26.98
Diversity	1.9 ± 0.1	2.2 ± 0.2	-2.57	0.014	-13.61
Biomass[a]	719 ± 153	640 ± 275	1.82	0.08	-16.2

[a] one pitfall plus one sweep net transect.

Abundance measured as individuals collected in 3 pitfalls/sweep-nets, richness as number of families collected in 3 pitfalls/sweep-nets, diversity through Shannon-Wiener diversity index, and biomass of arthropods as milligrams in 1 pitfall/sweep-net. Mean values ± SD, statistic (Z, GLMM-test), significance of the differences (P) and log-likelihood (LL) are given.

Comparisons between organic and conventional fields showed no significant differences for *Araneae*, *Coleoptera* or *Hemiptera* ($P > 0.41$ in all cases), higher numbers of *Acari*, *Collembola*, *Diptera*, *Hymenoptera*, and *Orthoptera* in organic fields ($P < 0.001$), and higher numbers of *Thysanoptera* in conventional fields ($P < 0.001$) (Fig. 2). Dominance analyses showed that *Collembola, Chloropidae* (*Diptera*), and *Aphididae* (*Hemiptera*) were dominant groups (Berger-

Parker index = 0.68 for these three groups together; respectively 0.76 and 0.58 in organic and conventional fields).
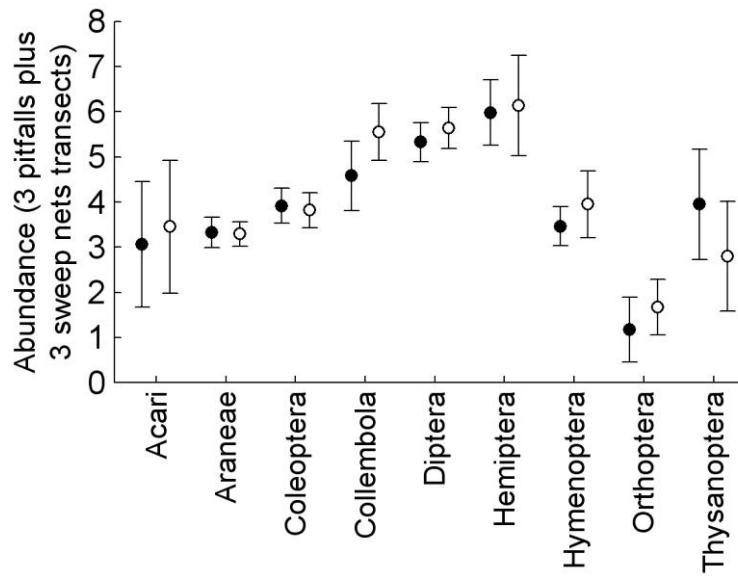
Figure 2. Numbers of individuals of the main arthropod orders sampled per field (three pitfalls plus three sweep nets transects), in organic (open circles) and conventional fields (black circles). Abundance measured as ln (number of individuals + 1). Means and SD values are given.

The result of the GLMMs showed that richness was higher in organic than in conventional fields (Table 5). After excluding dominant groups, richness was still higher in organic (40.3 ± 1.2) than in conventional fields (37.8 ± 1.1) (P = 0.02). No differences were found in family richness for *Araneae* and *Coleoptera* (respectively, P = 0.56 and P = 0.11, see list of families in Appendix C). For *Hemiptera*, richness was higher in organic fields (P = 0.01). As for *Diptera*, richness did not differ between organic and conventional fields (P = 0.95).

Diversity values were lower in organic than in conventional fields (Table 5). Within the most abundant orders, diversity was higher in in *Coleoptera* in organic fields (P = 0.01) and *Diptera* and *Hemiptera* in conventional fields (P < 0.001 in both), For *Araneae* we did not find statistical differences (P = 0.21). However, excluding dominant groups, organic fields showed higher diversity (respectively for organic and conventional fields: 2.7 ± 0.4 and 2.5 ± 0.3, P = 0.01). The

differences between diversity indices calculated including and excluding the dominant groups were higher for organic (0.8 ± 0.5) than for conventional fields (0.3 ± 0.4) ($P < 0.001$ in both cases).

The total estimated biomass of arthropods collected was slightly higher in organic fields than in conventional fields, although the difference was not significant (Table 5). By orders, only *Collembola* showed higher biomass in organic fields ($P < 0.001$), and *Thysanoptera* in conventional fields ($P = 0.002$). We searched for factors affecting arthropod biomass through GLMM. As weed richness was highly correlated with weed abundance (R = 0.72, $P < 0.001$) and weed diversity (R = 0.65, $P < 0.001$), weed richness was discarded from the plausible factors in GLMM, which included management type, weed abundance, weed diversity and weed cover as fixed factors, and field pair as random factor (Appendix D). Three models could be considered candidate models according to their differences in △AICc (<2). The variables included in the best model were management type, weed abundance, and weed diversity (Table 6), with 27.1% of the deviance explained (Appendix D). Model 1 differed from model 2 ($P = 0.03$), which also included weed cover (27.3% of the deviance explained). Model 3 included the interaction between management and weed diversity (27.3% of the deviance explained).

Table 6. Parameter estimates from the Generalized Linear Mixed Model with management (MA), weed abundance (WA) and weed diversity (WD) as factors affecting arthropod biomass.

Parameter	Estimate	SE	P
MA	4.091	0.172	< 0.001
WA	2.036	0.065	< 0.001
WD	3.153	0.139	< 0.001

DISCUSSION

In the dryland cereal agroecosystem studied, the first effect of organic farming was on the weeds, with knock-on effects (Hawes *et al.*, 2003) on the arthropods community, associated directly with this resource. Finally, the competition with weeds led into a decreased cereal production, as suggested by the lower number of cereal plants. The positive effect of a reduction in agrochemical applications on weed density has been experimentally demonstrated (e.g., Frampton and Dorne, 2007; Hyvönen and Salonen, 2002; Kleijn *et al.*, 2006). Weed and arthropod communities were also richer in organic fields and, in the case of weeds, more diverse than those of conventional fields. The average increases in weed abundance (202%), richness (176%), diversity (133%) and cover (421%) in organic fields were somewhat higher than those recorded in a dryland cereal area in northern Spain (Caballero-López *et al.*, 2010; José-María *et al.*, 2010; Romero *et al.*, 2008), and considerably higher than those reported for studies carried out at northern latitudes (e.g., Bengtsson *et al.*, 2005; Hole *et al.*, 2005; Moreby *et al.*, 1994). The higher development of weeds in absence of agrochemical treatment in these Spanish studies as compared to studies carried out at northern latitudes might be explained by several facts. First, the weed flora is more diverse in Mediterranean latitudes (Araújo *et al.*, 2007; Cowling *et al.*, 1996; Thompson, 2005). Second, in most Spain cereal is grown in a traditional two-year rotation system that creates a mosaic of ploughed, cereal and stubble patches, with some fallow fields left untilled for several years. Such system allows uncultivated fields to act as weed reservoirs from which their seeds may easily disperse, building up a rich weed community in organic cereal fields. In the more intensively cultivated cereal farmland in northern countries, these uncultivated weed reservoirs are less frequent, and thus the weed development in organic fields less marked. Third, in our study area fields are small (less than 2 ha) and field boundaries are narrow (mean width = 35 ± 25 cm, mean height = 40 ± 23 cm, n = 50, own data), favoring an easy exchange of seeds and arthropods among fields.

A limitation of our study could be that sampling was restricted to a single year of organic farming. However, rather than looking at an equilibrium situation, we were interested in knowing whether a quick response to organic treatment

could be observed. Some authors have noticed that rapid positive responses to agri-environmental measures would imply less costs, and that if an agri-environmental measure needs several years to become effective, perhaps it should not be implemented (e.g., Hole *et al.*, 2005). Moreover, the temperature and precipitation values of the study year were within half a standard deviation of the average for the last 30 years, suggesting that the results were probably not influenced by weather conditions. Finally, instead of performing several samplings through the spring, we restricted our sampling to just one time during May, due to the relatively short vegetative period in our study area. The sampling dates were selected to maximize the probability of collecting most weeds and arthropods, which in our study area have very short life cycles as compared to more northern latitudes. Besides, sampling effort for plants was adequate, since we sampled 80% and 89.8% of the species predicted by Clench equation, respectively in organic and conventional fields (Jiménez-Valverde *et al.*, 2003; Moreno and Halffter, 2001).

As in the study of Romero *et al.*, (2008), in our area *Lolium rigidum* (Gaudin) was the only dominant weed in conventional fields, due to its particular resistance to herbicides (Heap, 1997), and *Avena sterilis* (L.) and *Bromus diandrus* (Roth) were also relatively resistant. When herbicides were suppressed, a more complex weed community developed, and the prevalence of *L. rigidum* (Gaudin) decreased significantly, leaving space to other weeds, particularly broad-leaved species which are less resistant to the herbicides used (Kudsk and Streibig, 2003). Among these, several leguminous species were particularly important, since they contribute to nitrogen fixation, and thus to the development of a richer biocenosys. These species were *Vicia sativa* (L.), *V. spp.*, *Trifolium angustifolium* (L.) y *Ornithopus compressus* (L.), which together comprised ca. 7% of weeds in organic fields, as compared to only 0.2% in conventional fields. Some legumes are also related to increases in some arthropod groups as flower-consumers, chewing-herbivores and saprophages (Caballero-López *et al.*, 2010).

The best models selected by the GLMMs showed an influence of management type and cereal cover on weed richness, as well as an interaction between both variables. This means that as cereal cover decreased, the richness of the weed community increased, but only in the sample of organically managed

fields. Such relationship was not observed in conventional fields where herbicide treatment kept weeds under control. On average, organic farming implied a 24% reduction in the number of cereal plants. Assuming plant numbers are correlated with total cereal crop, organic farming also determined a similar decrease in agricultural production. Such a decrease is slightly higher than the 16.5% reported as mean variation among years in winter cereal production in Spain (MMAMRM, 2010).

As for arthropods, their abundance increased in organic fields compared to conventional fields (41%). Such increase is similar to those reported in previous studies (Bengtsson *et al.*, 2005; Frampton and Dorne, 2007; Hole *et al.*, 2005). The *Collembola, Chloropidae* (*Diptera*), and *Aphididae* (*Hemiptera*) were found to be dominant groups. These species were ca. 20% more abundant in organic fields than in conventional fields, concluding that their proliferation could be a direct consequence of the farming system. Clough *et al.*, (2007b) also found some dominant species of the *Staphylinidae* (*Coleoptera*) and Moreby *et al.*, (1994) found an increase of *Diptera* and *Aphids* (*Hemiptera*), the same orders identified as dominant in the present study. Their higher abundance and proliferation in organic fields could probably be favored by the greater cover in these fields of insect-pollinated weeds, particularly those with flowers, the typical niche of most of these insects. Arthropod richness was a 6.4% higher in organic fields. Most other studies have also recorded richness increases in organically managed fields (Clough *et al.*, 2007a; Hadjicharalampous *et al.*, 2002; Hole *et al.*, 2005; Pfiffner and Niggli, 1996), and the impact of organic management on arthropods has been interpreted as an indirect result of the impact of agro-chemical suppression on the vegetation (Siemann *et al.*, 1998). Finally, multivariate models showed that arthropod biomass was significantly influenced by farming practices, weed abundance and weed diversity. The best model explained only a 27.1% of the total deviance, which suggests that additional variables such as landscape complexity, distance to nearby organic fields, and field size could also be relevant (Clough *et al.*, 2007a; Concepción *et al.*, 2008).The lower arthropod diversity in organic fields is explained by the marked dominance in these fields of a few taxa, mainly *Collembola, Chloropidae* (*Diptera*), and *Aphididae* (*Hemiptera*). As argued by Shah *et al.*, (2003), who also found a higher diversity in conventional fields, the

Shannon-Wiener diversity index, despite its wide use in biodiversity studies, is particularly sensitive to changes in the abundance of dominant species in a sample. In their study, the diversity decrease in organic fields was due to the abundance of a dominant carabid, *Pterostichus melanarius* (Illiger). Several other studies also showed that organic management systems increased arthropod abundance and richness but not diversity (Booij, 1994; Clark, 1999; Hokkanen and Holopainen, 1986; Kromp, 1999). In our study, the greater abundance in organic fields of the three dominant groups mentioned above was probably related to a higher development of the weeds canopy, since *Chloropidae* adults are flower-consumers and chewing-herbivores, and *Aphididae* are suction-herbivores (Caballero *et al.*, 2010). Without considering these dominant groups, the frequency distribution of the remaining species indicated a significantly higher diversity in organic fields. This was consistent with richness values, which were higher in organic than in conventional fields.

Overall, our results confirm findings from previous studies, and suggest that organic farming may contribute to preserve biodiversity in the dryland cereal agroecosystem of our study area. Organic farming could thus be used as a way to minimize the negative impacts of agricultural intensification, and particularly to improve habitat quality for many vertebrate consumers such as several endangered steppe birds inhabiting dry cereal farmland in the Mediterranean region.

of the Dirección General de Investigación of the Spanish Ministry for Science and Innovation.

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Appendix A. Complete list of weed species identified, ordered by the frequency with which they were recorded in organically managed and conventional fields. Values are percentages of each species found in both field types.

Species	Organic fields	Conventional fields
Lolium rigidum (Gaudin)	50.3	80.2
Galium tricornutum (Dandy)	7.1	1.8
Bromus diandrus (Roth)	6.6	4.2
Anacyclus clavatus (Desf.)	4.2	0.7
Conyza canadensis (L.)	4.0	0.6
Raphanus raphanistrum (L.)	3.8	0.6
Avena sterilis (L.)	3.0	3.3
Vicia sativa (L.)	2.8	0.0
Polygonum aviculare (L.)	2.6	2.9
Filago pyramidata (L.)	1.7	0.1
Trifolium angustifolium (L.)	1.7	0.0
Filago lutescens (Jord.)	1.1	1.2
Vicia spp	0.9	0.1
Picnomon acarna (L.)	0.9	0.8
Lactuca serriola (L.)	0.8	0.2
Ornithopus compressus (L.)	0.7	0.1
Linaria viscosa (L.)	0.7	0.0
Spergula arvensis (L.)	0.7	0.0
Euphorbia serrata (L.)	0.6	0.0
Vicia ervilia (L.)	0.6	0.0
Filago gallica (L.)	0.6	0.0
Convolvulus arvensis (L.)	0.6	1.5
Anagallis arvensis (L.)	0.4	0.3
Carduus tenuiflorus (Curtis)	0.4	0.0
Hordeum murinum (L.)	0.4	0.1
Aegilops geniculata (Roth)	0.2	0.0
Andryala integrifolia (L.)	0.2	0.2
Ranunculus arvensis (L.)	0.2	0.0
Taeniatherum caput-medusae (L.) Nevski	0.2	0.3
Lathyrus sp	0.2	0.0
Anchusa azurea (Mill.)	0.2	0.0
Bromus squarrosus (L.)	0.2	0.0
Adonis aestivalis (L.)	0.1	0.0
Lupinus angustifolius (L.)	0.1	0.1
Centaurea cianus (L.)	0.1	0.0
Chenopodium album (L.)	0.1	0.2
Cnicus benedictus (L.)	0.1	0.1
Papaver rhoeas (L.)	0.1	0.1
Picris echioides (L.)	0.1	0.0

Species	Organic fields	Conventional fields
Torilis nodosa (L.)	0.1	0.0
Trifolium campestre (Schred. in Sturn)	0.1	0.0
Senecio vulgaris (L.)	0.1	0.1
Spergularia rubra (L.) J. Presl & C. Presl	0.1	0.0
Arabidopsis thaliana (L.) Heynh. in Holl & Heynh.	0.0	0.0
Ononis spinosa (L.)	0.0	0.0
Sherardia arvensis (L.)	0.0	0.0
Sonchus oleraceus (L.)	0.0	0.0
Taraxacum officinale (Weber)	0.0	0.0
Amaranthus albus (L.)	0.0	0.2
Cynodon dactylon (L.) Pers.	0.0	0.1
Rumex pulcher (L.)	0.0	0.1
Veronica hederifolia (L.)	0.0	0.2

Appendix B. Results of Generalized Linear Mixed Models (GLMMs) where management (MA) and cereal cover (CC) were factors affecting weed richness. Field pair was the random factor. The best models (1 and 2) were determined according to the lowest corrected Akaike's Information Criterion (AICc) and ANOVA test (P is given, when ΔAICc between one model and the best was less than two). The percentage of the explained deviance, degrees of freedom (d.f.) and model log-likelihood (LL) are also given

Model Number	AICc	△AICc	Explained deviance	d.f.	LL	P
1 CC + MA + CC*MA	82.3	0.00	56.8	5	-35.6	0.12
2 CC + MA	83.4	1.10	54.6	4	-37.4	< 0.001
3 MA	83.7	1.43	52.9	3	-38.8	
4 CC	169.6	87.39	0.8	3	-81.8	

Appendix C. Arthropod orders and families identified, and number of individuals collected in organic and conventional fields.

Order	Family	Organic fields	Conventional fields
Acari	*Gamasidae[a]*	2550	1639
	Oribatida[b]	23	4
Araneae	*Anyphaenidae*		2
	Atypidae	23	12
	Ctenizidae		1
	Dictynidae	1	
	Gamasidae	1	1
	Linyphiidae	467	530

113

Order	Family	Organic fields	Conventional fields
	Lycosidae	66	89
	Oonipidae	2	3
	Oxyopidae	17	17
	Palpimanidae	2	5
	Pholcidae		1
	Sicariidae		1
	Theraphosidae	2	
	Telemidae	1	
	Theraphosidae		1
	Theridiidae	31	17
	Titanoecidae		1
	Thomisidae	33	39
	Uloboridae		1
	Zoridae	77	50
	Zoropsidae	61	62
Coleoptera	*Aesalidae*		3
	Anthribidae	4	
	Anthicidae	59	25
	Brostrichidae	1	
	Bruchidae	2	
	Byrrhidae	1	
	Cantharidae	73	82
	Carabidae	287	305
	Cerambycidae	28	8
	Chrysomelidae	84	69
	Ciidae	1	
	Coccinelidae	162	153
	Curculionidae	119	145
	Dermestidae	15	12
	Dryopidae	5	1
	Elateridae	40	9
	Erotylidae		1
	Gyrinidae		2
	Histeridae		1
	Lampyridae	12	9
	Malachidae	3	4
	Meloidae	1	
	Nitidulidae	54	56
	Omaliinae		2
	Scarabeidae	40	19
	Scydmaenidae	2	1
	Staphylinidae	140	250
	Silphidae	204	287
	Silvanidae	1	2
	Tenebrionidae	2	
	Trogidae	2	7

Order	Family	Organic fields	Conventional fields
Collembola[b]	Collembola	9558	3630
Diplura	Campodeidae	9	70
	Japygidae	2	1
Diptera	Acroceridae	40	28
	Anthomiidae	1	
	Asilidae	41	93
	Bibionidae	5	2
	Camillidae	253	452
	Cecidomyiidae	534	365
	Ceratopogonidae	2	
	Chloropidae	6496	3147
	Conopidae	65	28
	Culicidae	38	67
	Dixidae		1
	Fanniidae	1	
	Heleomyzidae		2
	Hippoboscidae	75	49
	Lauxaniidae	11	23
	Lonchopteridae	2	
	Milichiidae		1
	Muscidae	183	277
	Mycethophilidae	2	
	Otitidae	3	1
	Phoridae	82	156
	Pipunculidae	229	242
	Platystomatidae	19	2
	Psilidae	21	40
	Ptychopteridae	18	13
	Sarcophagidae	2	
	Scathophagidae	92	105
	Scatopsidae	4	2
	Scenopinidae		1
	Sepsidae		8
	Sphaeroceridae	1	3
	Stratiomyidae	7	18
	Syrphidae	99	237
	Tabanidae	3	17
	Tachinidae		1
	Tethritidae	1	
	Therevidae		3
	Trichoceridae	112	301
	Trigonalidae	85	111
	Vermileonidae	55	52
	Xylophagidae		8
Embioptera	Oligotomidae		1

Order	Family	Organic fields	Conventional fields
Hemiptera	*Acanthosomidae*	28	5
	Alydidae	19	1
	Anthocoridae		2
	Aphididae	23014	13130
	Aphrophoridae	815	1674
	Cicadellidae	2	
	Cicadidae	247	261
	Cimicidae	1	6
	Delphacidae	2	
	Lygaeidae	15	5
	Miridae	44	1
	Nabidae	191	36
	Pentatomidae	56	2
	Pseudococcidae		1
	Psyllidae		6
	Reduviidae	12	1
	Rhopalidae	6	
	Scutelleridae	3	1
Hymenoptera	*Andrenidae*	1	
	Apidae	2	
	Cynipidae		2
	Evaniidae		4
	Formicidae	1863	867
	Pamphiliidae		1
	Pompilidae		2
	Sapygidae		4
	Siricidae	1	
	Trichogrammatidae		21
	Vespidae		2
	Xyelidae		3
Isopoda	*Philosciidae*	2	1
Lepidoptera	*Papilionidae*		3
	Pyralidae	2	1
Mecoptera	*Boreidae*		3
	Panorpidae	2	1
Miriapoda[c]	*Diplopoda[b]*	4	5
Neuroptera	*Ascalaphidae*	3	1
	Hemerobiidae	4	3
	Myrmeleonidae	3	2
Odonata	*Coenagrionidae*		1
Opinilionida	*Phalangiidae*	5	7
Orthoptera	*Acrididae*	45	36

Order	Family	Organic fields	Conventional fields
	Gryllidae	56	13
	Pamphagidae		2
	Tettigoniidae	25	20
	Trydactylidae	1	4
	Gryllotalpidae	30	19
Psocoptera	*Psocidae*		2
Siphonaptera	*Hystrichopsyllidae*	109	174
Thysanoptera	*Thripidae*	1015	2510
Thysanura	*Lepismatidae*	10	1
Total		50488	32334

[a] SubOrder

[b] Class

[c] SubPhylum

Appendix D. Results of Generalized Linear Mixed Models (GLMMs) where management (MA), weed abundance (WA), weed diversity (WD), and weed cover (WC) were factors affecting arthropod biomass. Field pair was the random factor. The best model was determined according to the lowest corrected Akaike's Information Criterion (AICc) and ANOVA test (P is given, when ΔAIC between one model and the best was less than two). The percentage of the explained deviance, degrees of freedom (d.f.) and model log-likelihood (LL) are also given.

Model number	AICc	△AICc	Explained deviance	d.f.	LL	*P*
1 MA + WA +WD	1684.6	0	27.1	5	-834.7	0.03
2 MA + WA +WD + WC	1685.6	0.98	27.3	6	-833.0	1
3 MA + WA +WD + MA*WD	1685.6	0.98	27.3	6	-833.2	
4 MA + WA +WD + WC + MA*WD	1687.0	2.40	27.4	7	-831.5	
5 MA + WD + MA*WD	1715.6	31.00	25.6	5	-849.9	
6 MA + WA + MA*WA	1785.6	101.00	21.1	5	-885.1	
7 MA + WA	1802.0	117.40	22.1	4	-894.8	

CAPÍTULO 4

Este capítulo reproduce íntegramente el siguiente artículo:

Ponce, C., Bravo, C., & Alonso, J.C. 2014. Effects of agri-environmental schemes on farmland birds: Do food availability measurements improve patterns obtained from simple habitat models?. *Ecology and Evolution* 4: 2834–2847.

CAPÍTULO 4

Effects of agri-environmental schemes on farmland birds: do food availability measurements improve patterns obtained from simple habitat models?

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ABSTRACT

Studies evaluating agri-environmental schemes (AES) usually focus on responses of single species or functional groups. Analyses are generally based on simple habitat measurements but ignore food availability and other important factors. This can limit our understanding of the ultimate causes determining the reactions of birds to AES. We investigated these issues in detail and throughout the main seasons of a bird's annual cycle (mating, post-fledging and wintering) in a dry cereal farmland in a Special Protection Area for farmland birds in central Spain. First, we modeled four bird response parameters (abundance, species richness, diversity and 'Species of European Conservation Concern' [SPEC]-score), using detailed food availability and vegetation structure measurements (*food models*). Second, we fitted new models, built using only substrate composition variables (*habitat models*). Whereas habitat models revealed that both, fields included and not included in the AES benefited birds, food models went a step further and included seed and arthropod biomass as important predictors, respectively in winter and during the postfledging season. The validation process showed that food models were on average 13% better (up to 20% in some variables) in predicting bird responses. However, the cost of obtaining data for food models was five times higher than for habitat models. This novel approach highlighted the importance of food availability-related causal processes involved in bird responses to AES, which remained undetected when using conventional substrate composition assessment models. Despite their higher costs, measurements of food availability add important details to interpret the reactions of the bird community to AES interventions, and thus facilitate evaluating the real efficiency of AES programs.

Keywords

INTRODUCTION

The demand of more food and biofuel (Tilman et al. 2011; Miyake et al. 2012) from modern agricultural activities has caused the decline of many species inhabiting farmland areas (Donald et al. 2001). Increased use of chemicals (pesticides, fertilizers, etc.), loss of noncropped habitats and loss of crop diversity are some of the most important factors affecting plant and animal populations in these ecosystems (Chamberlain et al. 2000; Vickery et al. 2001; Robinson and Sutherland 2002; Benton et al. 2003). However, agri-environmental schemes (AES, hereafter) are intended to reverse the environmental impacts of modern farming techniques on biodiversity (Stoate et al. 2009). It is generally accepted that an increase in habitat heterogeneity has a positive influence on biodiversity (Wuczyński et al. 2011). The European Union and the United States of America have spent several billion dollars in AES programs (Kleijn et al. 2006; Gabriel et al. 2010), but their effectiveness is still somehow questioned because different studies have reported contradictory results (Tscharntke et al. 2005; Kleijn et al. 2006). These differences may have been due to differences in the scale of study, with most clearly positive effects at local scales (Perkins et al. 2011) compared with larger scales (Verhulst et al. 2007; Davey et al. 2010), or in studies designed to enhance certain declining species (Wilson et al. 2009; Kleijn et al. 2011). AES are usually implemented at field scale, without controlling for the spatial complexity (vegetation structure and substrate diversity around managed fields) that affects the variables under study (Kleijn and Sutherland 2003; Tscharntke et al. 2005; Concepción et al. 2008; Gabriel et al. 2010; Winqvist et al. 2011). AES design and the research on their effectiveness usually focus on responses of just one or a few species (Breeuwer et al. 2009), although other species (MacDonald et al. 2012) or functional groups (granivorous, insectivorous, etc.; Henderson et al. 2000) may also obtain benefits. AES studies rarely analyse the responses of the whole bird community, ignoring that biodiversity maintenance should be a priority (Ekroos et al. 2014). Finally, although habitat and feeding requirements of species change through the year (Marfil-Daza et al. 2013), most AES studies evaluate effectiveness during a single season.

In the present study, we investigated the effects of an AES on a steppe bird community in a dry cereal farmland area in central Spain. We did this by analyzing various abundance and diversity parameters, which define direct bird responses of the farmland bird community to the AES (Díaz et al. 2006). The populations of 17 dry farmland bird species present in Spain are rapidly declining (Escandell et al. 2011), even faster here than in other European countries (EBCC 2010). Despite this, Spain still holds the most important breeding populations of several species classified as endangered at a continentwide scale, for example, the pin-tailed sandgrouse (*Pterocles alchata*), lesser kestrel (*Falco naumanni*) or great bustard (*Otis tarda*). Also, Spain holds significant wintering populations of common European species like the meadow pipit (*Anthus pratensis*) or the skylark (*Alauda arvensis*). Thus, Spain has the highest impact on the European Farmland Bird Index (EFBI), an indicator for biodiversity health on farmland areas (Butler et al. 2010a,b). As the scheme prescriptions and measures of our AES (Table 1) were designed for a broad spectrum of bird species, the results of this study may be considered as widely applicable for managers and ecologists in general.

Table 1. List of field types and prescriptions of the agri-environmental scheme (AES) implemented in the study area

Field type	Short name	Origin	Scheme prescriptions
Legume	LegAES	AES	Growing organic legumes. Sowing Seed on ploughed fields (190kg/ha) in October with a mixture of up to 20% cereal seed. No use of dressed seed. No agricultural activities (weed and arthropod control, tillage tasks, fertilizer applications) from sowing to harvest after 10 July[1]
	LegNAT	Natural, non-AES farming	Not applicable
Cereal stubble	CerStubbleAES	AES	Maintenance of cereal stubble. No agricultural activities from usual harvest date from June-July to 31 December and from 1 April to 1 July. Tillage tasks allowed from 1st January to 31st March without use of herbicides or insecticides
	CerStubbleNAT	Natural, non-AES farming	Not applicable
Fallow	FallowAES	AES	Interruption of the cereal production for $\geq$ 1 years. No agricultural activities (weed and arthropod control, tillage tasks, fertilizer applications) are allowed from July to the next July. The agreement can be renewed annually. Fields included in this AES must have been used for agricultural purposes on the last three years
	FallowNAT	Natural, non-AES farming	Not applicable
Legume stubble	LegStubbleAES	AES	No scheme prescriptions. It comes from the LegAES measure
	LegStubbleNAT	Natural, non-AES farming	Not applicable
Cereal crop	CerNAT	Natural, non-AES farming	Not applicable
Plough	Plough	Natural, non-AES farming	Not applicable
Plough with sprouted weeds	Plough2	Natural, non-AES farming	Not applicable
Edge	Edge	Natural, non-AES farming	Not applicable

[1] Although AES limit the harvest date from 10 July, we accepted farmers harvesting legume fields earlier (but always after 31 June) to feed sheep or to collect the seed for the following sowing season

The AES was funded by a program of preventive, corrective, and compensatory measures to balance the impact of the M-50 and R-2 highways on the population of great bustards and other steppe-land birds in the Important Bird Area (IBA) Talamanca- Camarma and the Site of Community Importance cuenca de los ríos Jarama y Henares. The two highways were built in the inner border of the Special Protection Area (SPA) 139 Estepas cerealistas de los ríos Jarama y Henares, which is included in the IBA. To implement the AES, first we contacted with agricultural agents to prepare a meeting with farmers from the SPA. Second, we evaluated all fields farmers offered to be included in the AES following suitability criteria such as distance to power lines, fences, towns, etc. Third, once a field was accepted and AES measure implemented on it, we made periodic checks of each field. Finally, payments to farmers were performed by the company operating the highways.

However, our main purpose was not just to test the AES effectiveness. Our first major objective was to explore the effects of a differing quality of the field data commonly used to investigate bird responses to AES. We did this by comparing the predictive capacity of response models based on simple habitat measurements (called *habitat models* hereafter) with that of models also based on habitat variables and much more detailed food availability and vegetation structure measurements (*food models* hereafter). We also analyzed the cost in terms of money and effort spent on each set of models, because it is expected that the amount of time and funds invested should correspond to the quality of the results obtained. Most AES effectiveness studies have been carried out with relatively low investment in field work, often using the composition of substrates selected by the birds before and after AES implementation (see review in Kleijn and Sutherland 2003). The only conclusion that can be drawn from these studies is the positive, neutral or negative effect of the AES on individual behavior, whereas in most cases the ultimate causes, processes and population level consequences remain largely unknown. However, it has been suggested that the association between bird species and their habitat is determined by the quantity and quality of the resources provided (functional space available to a species), not only the habitat per se (Boyce and McDonald 1999; Butler and Norris 2013). For example, it is currently admitted that agriculture intensification has determined massive declines of

farmland bird species, and these declines are due to different processes as reduced food resources (food availability hypothesis, Newton 2004), reduced refuge quality (refuge and nesting hypothesis; Benton et al. 2003), or both in some cases (Campbell et al. 1997; Butler et al. 2007, 2010a,b). Some studies on these questions are species-specific and do not consider the whole bird community (Breeuwer et al. 2009; Bretagnolle et al. 2011). However, it is considered that biodiversity loss can compromise many ecosystem services (Cardinale et al. 2012) and the impacts of species loss on primary productivity are comparable with impacts from climate warming (Hooper et al. 2012; Tilman et al. 2012).

Our second main objective was to compare different bird responses: abundance, richness, diversity and SPECscore, an index based on the Species of European Conservation Concern (BirdLife International 2004) among three periods of the annual cycle: wintering, mating, and postfledging. Most studies have explored ways to enhance breeding success, but very few have been carried out during the wintering or postfledging seasons, which are also critical for birds. It is known that ecological circumstances during the nonbreeding seasons (postfledging and wintering) may affect body condition and survival rates (Siriwardena et al. 2000; Stoate et al. 2004), and influence the dynamics of the population (Siriwardena et al. 2007; Butler et al. 2010a,b). Comparisons among seasonal models enabled us to investigate in detail the processes involved in bird responses to AES, that is, which aspects of birds' requirements are better fulfilled by the agri-environmental measures, and which part of their annual cycle is more influenced by these measures. To our knowledge, this is the first study comparing the predictive power of models using field variables of differing quality and exploring the responses of the whole bird community in different seasons.

Our main study hypotheses were that (i) as birds requirements differ throughout different periods of their annual cycle, agri-environmental schemes can lead to different effects in different seasons, and (ii) improving food availability measurements should lead to significantly higher predictive power than just using simple habitat measurements, which sometimes may compensate for the higher field work costs incurred.

MATERIAL AND METHODS

Study area, field types, and agri-environmental measures

The field work was carried out in the SPA 139 Estepas cerealistas de los ríos Jarama y Henares, located in Madrid region (central Spain), where an AES has been running since 2003. Specifically, we sampled four sites within this area (Fig. 1). The region has dry cereal cultivation as its main land use, and all the sites share major environmental–climatic, biogeographic conditions, as well as a similar steppe bird community.
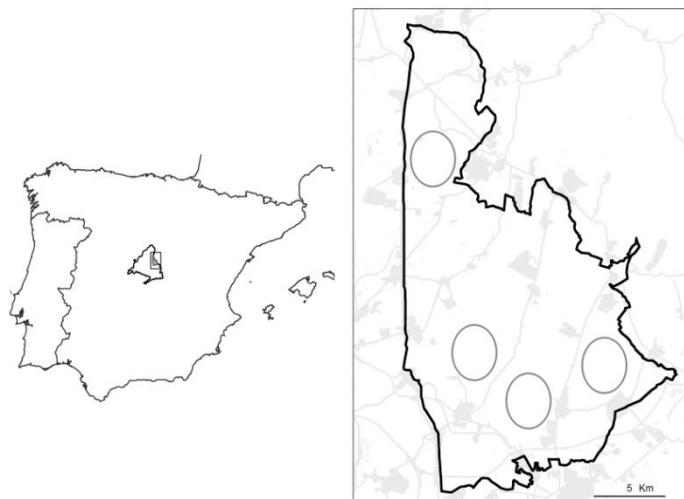
Figure 1. Location of the study area in the Iberian Peninsula. The figure at the right shows the SPA 139 Estepas Cerealistas de los Ríos Jarama y Henares and the four sites (ellipses) where field work was carried out.

The study area was a typical non-intensive farmland area, with small fields, margins between neighbour and the presence of legumes and cereal stubbles (managed differently from the AES, with the use of pesticides) and fallow fields. Most cereal was grown following a traditional 2-year rotation system (fields are cultivated every second year). We mapped habitat types on GIS-based maps along the transects (200 X 500 m) to calculate the surface of land uses on the same day when bird censuses were carried out. We defined eight field types (Table 1). The agri-environmental measures implemented were maintenance of cereal stubble, growing legumes organically (vetch *Vicia sativa*), and interruption of the cereal

production (fallows; Table 1). The managed surface for transects where birds were censused ranged from 0% to 76%. We measured the width of 50 margins at random locations and considered this measure constant throughout the study (Lane et al. 1999).

Bird surveys

We censused birds from 2006 to 2009 in the winter (between 15 and 31 December), mating (between late April and early May) and postfledging seasons (early July). We carried out one census per season and site (four censuses in each season), which is enough to get reliable information about habitat selection and the above-mentioned parameters (Hanspach et al. 2011). The observer (CP) walked along 9–14 linear transects per site, each 500 m long and 100 m wide at each side of the path (totaling 660 transects (220 in each of the three periods – wintering, mating and post-fledging–) and more than 24,000 birds (see Appendix S1 in Supporting information). Within each site, a distance of 100 m was kept between the end of a transect and the beginning of the next transect. Transects within each site were located at paths that are only used by sheep and farmers. The path within each site was circular and the observer avoided double counting by not considering birds near the beginning and end of this path and looking where flying birds landed during each survey. There was no spatial correlation between consecutive transects in a site (Appendix S2). Birds flying over transects were only taken into account if they were clearly using the field (hunting, hovering, etc.). For each bird flock spotted, we recorded its species composition, number of individuals, and field type where it landed.

Food availability

Abundance of arthropods, seeds, and vegetation were sampled in three fields of each substrate type, in the four study sites in the SPA from spring 2006 to spring 2008 (three mating seasons, two postfledging seasons and two wintering seasons) to obtain average representative values for each substrate in each season and site. Arthropods were sampled visually following the same methodology as Lane et al. (1999). Two observers walked at low speed (120 m/h, to avoid effects of

detectability due to vegetation characteristics) along linear transects (30 m long and 0.5 m wide at both sides) in the middle of each field. We counted and identified every specimen to the most accurate taxonomic level possible by means of visible characters (Oliver and Beattie 1993). We collected a random sample of 7515 arthropods, 12% of the those detected) and assessed the mean biomass of each group using linear regression of weight as function of body length, which was estimated for each individual during the transect (Hódar 1996; Ponce et al. 2011). We measured all collected arthropods in the laboratory. The mean lengths obtained allowed us to estimate the length of the observed and noncollected specimens and to assess the biomass. Finally, for each site, transect, and season, we calculated the surface of each field type and so obtained a weighted average biomass per hectare.

Vegetation abundance and structure were assessed throwing a metal square (25 X 25 cm) at 20 random locations in the middle of each field. Data recorded were total plant cover (%) and height (minimum, maximum, mean and most frequent, in cm). In these samples, we evaluated the roughness of the ground (the degree of flatness of each field) using three categories (low, medium, or high). We also estimated visually the total numbers of seeds on plants and on the ground classifying them in four size categories based on their maximum length (<1 mm, 1–5 mm, 5–10 mm, and >10 mm). We spent the time necessary to count each seed, regardless of the vegetation present or the field type. We collected a sample of seeds to measure, weigh, and estimate the mean biomass for each size in the laboratory. Finally, for each site and season, we calculated the average seed biomass in each substrate type.

The costs for habitat and food models were calculated according to the money and time required to obtain the data. We considered field and laboratory work for food models but only field work for habitat models. Money spent in food models included costs of sampling and weighing seeds and arthropods, measuring vegetation structure and recording field uses. In the case of habitat models, the costs were those of mapping habitat land uses. Both models shared identical bird census costs. We estimated the costs considering salaries, travel, and daily subsistence allowances of observers, and field and laboratory material (excluding

those provided by the research center). Time (h) needed for each set of models was calculated according to the effort required for obtaining field and laboratory data.

Statistical analyses

The response variables were total bird abundance, richness (number of species), Shannon–Wiener diversity index and SPEC-score in each transect, season, and site. The SPEC (Species of European Conservation Concern) is the conservation status of all wild birds in Europe (BirdLife International 2004). The SPEC-score index is important because it gives more importance to species of major concern in Europe. Bird species are classified into five categories (from SPEC 1 to SPEC 3 plus Non-SPEC and Non-SPEC[E] (previously SPEC 4, not detected during the surveys). For the present study, the highest value (4) was assigned to SPEC 1 species (major concern) and the lowest (0) to Non-SPEC species (least concern). The SPEC score was the sum of these values for each different species detected in each transect.

We built averaged mixed models to analyze the response variables. To select the fixed factors in the model, we firstly performed a Principal Component Analysis (PCA), with the Varimax Normalized factor rotation, with all plausible variables in each season (Appendix S3) to explore the degree of association among variables. The percents (%) of field types and vegetation cover were arcsine square root transformed. We only considered axes with eigenvalue >2. We selected the variable that correlated most strongly with the axis for further analyses (always ≥0.7, Appendix S4) to reduce multicollinearity among variables (Barrientos and Arroyo 2014). We preferred to use raw variables instead of PCA factors because their meaning is easier to interpret (Barrientos 2010). This technique allows highly correlated variables to be discarded (which can also be done with simple correlations) and objectively selects the most biologically meaningful and influential variable with each factor (Barrientos 2010). Secondly, we followed the procedure described in the study of Zuur et al. (2009) to select the random factor in mixed models. We built the most complex model (beyond optimal model) with all fixed factors from PCA and including different random factors. We considered year, site, both combined, or year nested within site as plausible random factors. To select the random factor, we used the results from the ANOVA

test in R-program (R Core Team 2013). We built all possible models for the four response variables in each season using a subsample of the data (154 cases for each season, 70% of the dataset), leaving the rest of data for the validation process. We selected models with an increase in corrected Akaike0s Information Criterion (ΔAICc) over the best model <5 as candidate models (Burnham and Anderson 2002). Finally, with all these models, we performed an average model estimation, with the package MumIN (Barton 2013) in Rprogram, in which the parameter estimates of all models were combined (Burnham and Anderson 2002). The random factor was that previously selected and the error structure was Poisson for abundance, richness and SPEC-score and Gaussian for diversity. The final averaged models included those variables identified as significant (those whose confident interval excluded 0 value; Alonso et al. 2012; Delgado et al. 2013).

We developed two sets of models for each response variable. The first was that of *food models* which used detailed seed and arthropod biomass measurements and parameters describing the vegetation structure and the surfaces of each field type as candidate variables (see Appendix S3). The second was that of *habitat models*, built using field types and surfaces (Appendix S3). In all cases, the variables selected from each axis had a correlation value ≥0.7 (Appendix S4). We explored the predictive power of each set of models on the 30% previously discarded data set (66 transects distributed evenly among all study sites in each season). This validation shows how accurately the best model predicts data not used before (Seiler 2005; Vaughan and Ormerod 2005). In spite of the acknowledged importance of model validation in behavioral and ecological studies in general, and distribution modeling studies in particular, this issue has been generally ignored in the literature analyzing the efficiency of AES programs.

RESULTS

Wintering season

In winter food models, bird abundance was positively predicted by seed biomass found in AES legume fields, surface of AES fallows and ploughed fields, and surface of non-AES cereal stubbles (Table 2). The surface of AES fallows was the most

important variable. Bird richness was determined by AES fallow surface, total arthropod biomass, and seed biomass from non-AES high-quality fields (stubbles, fallows, and legume fields; Table 2). Again, the most important variable was the surface of AES fallows. Bird diversity was best explained by a model including the surface of ploughed and cereal stubble fields from regular farming activity (notice: in this case the influence was negative), and the seed biomass from non-AES high-quality fields (non-AES stubbles, fallows, and legumes; Table 2). The SPEC score was determined by seed biomass in legume fields and surface of fallows, in both cases from AES. The standardized regression coefficients showed that AES fallow surface was the most important variable predicting SPEC-score.

Table 2. Model-averaged estimates of the food models for bird abundance, richness, diversity and SPEC-score during the wintering, mating and post- fledging periods. The statistics given are: sum of Akaike weights of the models in which the predictor was retained (Σ), parameter estimate of the regression equation (b), standard deviation of the regression parameter (SE), lower and upper confident limits of b, and standardized coefficients of predictors (β). Non-significant predictors are not included. Factors are ordered by magnitude of the β coefficient.

Period	Variable	Parameter	Σ	b	SE	Lower CI	Upper CI	β
Wintering	Abundance	Intercept		3.69	0.24	3.22	4.16	0.08
		FallowAES	1	4.36	0.46	3.44	5.28	0.18
		SeedLegAES	1	1.66	0.63	0.40	2.92	0.10
		CerStubbleNAT	1	0.88	0.17	0.54	1.22	0.01
		Plough	0.88	0.43	0.21	0.01	0.85	0.01
	Richness	Intercept		1.27	0.14	0.99	1.55	0.14
		FallowAES	0.87	0.99	0.31	0.37	1.61	0.25
		SeedHQFNAT	1	0.54	0.18	0.18	0.90	0.08
		ArthrTot	0.68	0.19	0.07	0.05	0.33	0.01
	Diversity	Intercept		0.26	0.05	0.15	0.37	0.00
		Plough	1	1.03	0.08	0.87	1.20	0.27
		CerStubbleNAT	1	-0.47	0.10	-0.68	-0.27	-0.16
		SeedHQFNAT	1	0.28	0.09	0.10	0.46	0.08
	SPEC-score	Intercept		2.54	0.16	2.22	2.85	0.10
		FallowAES	1	2.71	0.29	2.13	3.28	0.19
		SeedLegAES	1	1.30	0.32	0.66	1.94	0.10
Mating	Abundance	Intercept		2.22	0.08	2.06	2.38	0.01

Period	Variable	Parameter	Σ	b	SE	Lower CI	Upper CI	β
		CerStubbleAES	1	2.64	0.49	1.66	3.62	0.11
		LegAES	1	1.84	0.52	0.80	2.88	0.08
		FallowNAT	1	1.36	0.45	0.47	2.25	0.05
		Plough2	1	0.85	0.38	0.09	1.62	0.03
	Richness	Intercept		2.30	0.15	2.01	2.59	0.19
		LegAES	1	1.69	0.12	1.44	1.94	0.12
		FallowNAT	1	1.30	0.16	0.99	1.61	0.11
		LegNAT	0.76	0.69	0.18	0.34	1.05	0.07
	Diversity	Intercept		0.35	0.02	0.30	0.40	0.04
		FallowNAT	1	0.41	0.09	0.22	0.60	0.19
		LegNAT	0.89	0.28	0.09	0.10	0.47	0.12
	SPEC-score	Intercept		2.49	0.20	2.08	2.90	0.08
		FallowNAT	1	0.95	0.40	0.15	1.75	0.06
		Plough2	0.94	0.70	0.34	0.00	1.37	0.04
Post-fledging	Abundance	Intercept		3.20	0.39	2.42	3.98	0.04
		ArthrFallowNAT	0.85	1.93	0.86	0.21	3.65	0.06
		Plough	0.90	-1.12	0.37	-1.86	-0.38	-0.01
		LegStubbleAES	0.75	0.74	0.35	0.04	1.44	0.01
	Richness	Intercept		1.07	0.21	0.65	1.49	0.13
		FallowAES	1	1.26	0.18	0.91	1.62	0.13
		ArthrFallowNAT	0.94	0.67	0.29	0.08	1.26	0.11
		Plough2	1	0.60	0.22	0.16	1.03	0.07
	Diversity	Intercept		0.23	0.04	0.16	0.30	0.03
		FallowAES	0.89	0.30	0.10	0.09	0.50	0.12
		Plough2	0.88	0.31	0.10	0.12	0.50	0.12
		ArthrFallowNAT	0.65	0.24	0.11	0.02	0.46	0.11
	SPEC-score	Intercept		1.38	0.15	1.09	1.68	0.06
		Plough2	1	1.50	0.57	0.36	2.64	0.24
		ArthrFallowNAT	1	0.74	0.32	0.11	1.37	0.07
		FallowAES	0.65	0.55	0.15	0.25	0.85	0.02

The habitat models for bird abundance and richness included the surface of AES legumes and fallows as predictors, while models for richness and diversity included the surface of non-AES fallows (Table 3). Bird abundance and SPEC score

were predicted by the same variables. Also, non-AES cereal stubbles had the lowest effect. The surface of non-AES fallows had a positive effect on bird richness and diversity. Diversity was also positively affected by the surface of ploughed fields (Table 3). Natural or managed fallows had the highest standardized coefficient in all habitat models.

Table 3. Model-averaged estimates of the habitat models for bird abundance, richness, diversity and SPEC-score during the wintering, mating and post-fledging periods. The statistics given are: sum of Akaike weights of the models in which the predictor was retained (Σ), parameter estimate of the regression equation (b), standard deviation of the regression parameter (SE), lower and upper confident limits of b, and standardized coefficients of predictors (β). Non-significant predictors are not included.

Period	Variable	Parameter	Σ	b	SE	Lower CI	Upper CI	β
Wintering	Abundance	Intercept		4.70	0.13	4.45	4.96	0.05
		FallowAES	1	2.27	0.56	1.14	3.39	0.12
		LegAES	1	1.67	0.25	1.18	2.16	0.04
		CerStubbleNAT	0.87	0.70	0.06	0.58	0.82	3.84E-03
	Richness	Intercept		1.27	0.25	0.77	1.78	0.26
		FallowAES	1	0.68	0.29	0.10	1.27	0.16
		LegAES	1	0.63	0.23	0.17	1.10	0.12
		FallowNAT	1	0.56	0.26	0.04	1.07	0.12
	Diversity	Intercept		0.21	0.08	0.04	0.38	0.06
		FallowNAT	1	0.28	0.12	0.04	0.53	0.11
		Plough	1	0.25	0.11	0.03	0.47	0.09
	SPEC-score	Intercept		2.47	0.17	2.13	2.81	0.10
		LegAES	1	1.00	0.19	0.62	1.37	0.05
		FallowAES	1	0.59	0.26	0.07	1.11	0.04
		CerStubbleNAT	1	0.19	0.08	0.03	0.35	3.78E-03
Mating[a]								
	SPEC-score	Intercept		1.80	0.18	1.44	2.16	0.05
		LegNAT	1	1.40	0.50	0.40	2.41	0.11
		Plough	1	-0.52	0.14	-0.81	-0.23	-0.01
		FallowNAT	1	0.39	0.19	0.02	0.77	0.01
Post-fledging								
	Abundance	Intercept		1.80	0.18	1.44	2.16	0.01
		FallowNAT	1	1.73	0.52	0.70	2.77	0.03

Period	Variable	Parameter	Σ	b	SE	Lower CI	Upper CI	β
	Richness	Intercept		0.88	0.11	0.67	1.09	0.05
		FallowAES	1	0.97	0.27	0.42	1.51	0.15
		FallowNAT	1	0.62	0.29	0.04	1.20	0.10
		Plough2	0.75	0.59	0.25	0.09	1.09	0.08
	Diversity	Intercept		0.22	0.04	0.14	0.29	0.03
		FallowAES	1	0.29	0.10	0.09	0.50	0.12
		Plough2	1	0.24	0.11	0.02	0.46	0.11
		FallowNAT	1	0.20	0.09	0.03	0.37	0.07
	SPEC-score	Intercept		1.37	0.14	1.08	1.66	0.06
		FallowNAT	1	0.82	0.26	0.30	1.34	0.06
		Plough2	1	0.57	0.20	0.17	0.98	0.03

[a] Food and habitat models retained the same variables

Mating season

During the mating season, four significant variables were retained in food models for bird abundance and three for richness, and two diversity and SPEC score (Table 2). The four variables influencing bird abundance had positive effects. Most important were the surfaces of cereal stubbles and legume fields from AES, followed by the surfaces of non-AES fallows and ploughed fields with sprouted weeds.

The best food models for bird richness and diversity included non-AES fallow and non-AES legume surfaces, both showing similar importance (Table 2). The model for bird richness also included AES legume fields as the most influential variable. The averaged model best explaining SPEC score included surfaces of non-AES fallows and ploughed fields with weeds (Table 2). The influence of both variables was positive, fallow surface showing the largest effect.

Habitat models best explaining bird abundance, richness, and diversity during the mating season retained the same predictors as food models (Table 3). The model averaging process showed that three variables influenced SPEC score: surface of non-AES fallows and legumes with positive effects, and surface of ploughed fields, with slightly negative effects.

Postfledging season

The biomass of arthropods in non-AES fallows was included in final food models for abundance, richness, diversity and SPEC-score (Table 2). The surface of ploughed fields with sprouted weeds was also included in the SPEC-score model with a higher regression coefficient value. Also, the final model for bird abundance included a negative effect of ploughed surface. The final models explaining bird richness and diversity shared all predictors, namely arthropod biomass in non-AES fallows, surface of AES fallows, and surface of ploughed fields with sprouted weeds. In all cases, the amount of AES fallows was the most important variable (Table 2).

Non-AES fallow surface was present in all final habitat models and ploughed land with sprouted weeds was absent only in the model for bird abundance (Table 3). The surface of AES fallows was retained in final models for bird richness and diversity (Table 3). All variables had a positive influence, and those derived from AES showed the highest importance when they were included in the models.

Comparison of models and their cost

The sensitivity analysis showed that food models had a consistently higher predictive ability than habitat models. The average increase in fit to the data was 13%, reaching a 20% in some variables (Table 4). Fit values were highest for bird abundance and lowest for SPEC-score in both, food and habitat models, and in all three seasons. SPEC score models showed the highest differences in predictive ability between food and habitat models (18% on average). Differences between food and habitat models were usually higher during the postfledging season than during the wintering season.

Table 4. Sensitivity analyses for testing the predictive ability of the food and habitat models

Season	Dependent variable	Model predictive ability (%)		
		Food	Habitat	Relative increase [1]
Wintering	Abundance	61.8	54.8	12.8
	Richness	56.7	49.3	15.0
	Diversity	50.3	44.9	12.0
	SPEC-score	41.5	34.6	19.9
Mating	Abundance	61.4	-[2]	-
	Richness	52.7	-[2]	-
	Diversity	42.9	-[2]	-
	SPEC-score	39.8	34.6	15.0
Post- fledging	Abundance	51.4	42.4	21.2
	Richness	50.3	41.8	20.3
	Diversity	43.5	36.3	19.8
	SPEC-score	38.4	32.3	18.9

SPEC, Species of European Conservation Concern.

[a] Calculated as (food / habitat) x 100

[b] Food and habitat models retained the same variables

Obtaining food models required almost 2800 h, which was 20 times more than those needed for habitat models (Table 5). Most of this extra time was due to field work (86%). Also, food models needed more than 16,000 € mainly due to the salaries (65%), a cost five times higher than that of habitat models.

Table 5. Comparison of costs (in €, work hours or number of people) incurred to measure variables used in food and habitat models

Effort	Food models	Habitat models
People (n)	2	1
Field time (h)	2422	135
Laboratory time (h)	372	0
Total time (h)	2794	135
Total time (days)	276	45
Salaries (€)	10750	2250
Fuel (€)	1830	675
Food (€)	2440	450
Field material [a] (€)	1286	0
Laboratory material [b] (€)	105	0
Total cost (€)	16411	3375

[a] Small sampling equipment only

[b] Small laboratory equipment only. A binocular loupe and two optical microscopes were provided by the research institute

DISCUSSION

Food versus habitat models

Habitat and food averaged models retained similar predictor variables, and the main conclusion from both sets of models was that AES benefited steppe birds by increasing the responses analyzed. However, food models were more effective in explaining bird responses, going beyond the simple assessment that AES measures were favorable. First, at least in winter and during the post-fledging season, they had a higher predictive power than habitat models (respectively, 15% and 20% higher). Second, they helped inferring important details about the ultimate causes underlying bird responses in different periods of the annual cycle. For example, while during winter habitat models included the surface of fallows, stubbles and legume fields as main predictors, food models revealed the specific importance of seed biomass for most of the response variables. Food models also highlighted the importance of arthropod biomass in fallows during the post-fledging season, when invertebrates are a major component of the diet of juveniles in most bird species.

These results show that in two of three seasons birds responded primarily to the amount of food rather than to the surface of fields or the vegetation structure, which was not included in any model. These results support our hypothesis that increasing food resources leads to significantly higher birds numbers for those periods in our study area.

An estimate of the seed biomass, either only in legume fields or altogether in the three substrates considered of high quality (legume fields, stubbles and fallows), was retained in the best winter food models for all response variables investigated. Food models thus captured the important role that seeds play as a source of energy and nitrogen for steppe birds during winter (Evans *et al.*, 2011). The amount of seeds in AES legume fields was particularly important for abundance and SPEC-score models. The positive effect observed on averaged model for the SPEC score means that by increasing the offer of legume fields in winter, the AES program contributed to enhance not only the overall abundance of birds maybe by attracting of local birds to food, but particularly that of species with special conservation interest. This result contradicts the findings og Kleijn *et al.* (2006), who suggested than endangered species rarely benefited from AEM. Richness and diversity of the bird community responded to the total seed biomass in all high-quality substrates (fallows, cereal stubbles, and legume fields) including non AES fields.

Food models identified the biomass of arthropods as a further variable influencing species richness during winter. That the richness of the steppe bird community was affected by arthropod biomass in winter was a quite unexpected result, as during this season most birds are typically granivorous. This result highlights the importance that arthropods may have even in winter for the bird community of dry cereal farmland in Mediterranean latitudes, whose climatic conditions may favor the presence of arthropod reservoirs available for wintering bird species.

In sum, with the exception of the mating season food models were better than habitat models in predicting bird community responses. In the latter, the direct importance of seeds and arthropods would have gone unnoticed. The gain in predictive power was highest for bird abundance and richness models in summer

(respectively, 21.2% and 20.3%), and the SPEC-score model in winter (19.9%). The advantage of a higher predictive power of food models should, however, be balanced against their much higher cost. In this study, the cost of obtaining data for food models was five times higher than for habitat models. Two persons, 130 days field work and 146 days laboratory work (ca. 3000 working hours in total) were necessary to measure the biomass of arthropods and seeds, and the vegetation structure variables. In addition, fuel, materials, and maintenance costs of personnel were also higher in food models. A five times higher cost could appear to be an excessive expenditure, but the additional cost of quantifying food availability may only represent a minor fraction of the total cost of agri-environment programmes.

Our study calls attention to the fact that bird responses could remain unexplained if they are judged only from an assessment of the habitat variables. This could lead to erroneous AES efficiency assessments. In contrast, measurements of food availability and vegetation structure could add important details to help interpreting the reactions of the bird community to AES interventions and thus facilitate evaluating the real efficiency of AES programs. The decision whether to invest in such detailed measurements should be taken considering the specific circumstances of each particular AES program.

Benefits of AES measures in different seasons

Considering all seasons together, the most effective AES measure was probably the provision and maintenance of fallows. Fallow fields were among the most significant predictors in a majority of food and habitat models in the three seasons. Previous studies already highlighted the importance of fallows in providing food and refuge for several steppe bird species (Duelli and Obrist 2003; Suárez *et al.* 2004; Billeter *et al.* 2008). Fallows are perhaps the only substrate offering sufficient amount of varied food types including weeds, seeds, and arthropods (Campbell *et al.* 1997; Herkert 2009; Lapiedra *et al.* 2011). It is therefore not surprising that these substrates appear in many AES studies as critical to increase bird abundance, richness and diversity.

Maintenance of cereal stubbles through the winter did not appear to be an AES measure providing a significant benefit to steppe birds, probably because in

nonintensive dry cereal areas non-AES cereal stubbles are already abundant in winter. A previous study (Suárez et al. 2004) also suggested that in Spain stubble maintenance through the winter did not benefit farmland birds since these can feed on various non-cultivated substrates. However, in areas where farming is more intensive and thus cereal stubbles scarce or are usually absent in winter, an AES including cereal stubble maintenance may certainly benefit steppebirds (Gillings et al. 2005; Concepción and Díaz 2011; Concepción et al. 2012). In winter food models, the surface of natural cereal stubbles had a positive effect on bird abundance, but a negative effect on bird diversity. This could be so due to the differences in diet among species (Princé et al. 2012), or simply because a marked preponderance of a single species as skylark (*Alauda arvensis*, Appendix S1) implies a reduction of species diversity values. Anyway, raising habitat quality by increasing the amount of food in winter may also produce delayed benefits during the following breeding season (Gillings et al. 2005). Several studies showed that breeding success or fitness of some species were correlated with conditions experienced during the preceding winter (Peach et al. 1999, 2001; Siriwardena et al. 1999, 2000, 2007).

The positive effect of ploughed fields on winter bird abundance and diversity was surprising, given the low values for vegetation cover and arthropod and seed biomass typical of these substrates (this study; see also Díaz and Tellería 1994). However, other authors qualified ploughed fields as important for some bird species (Suárez et al. 2004). For example, wagtails (*Motacilla spp.*) or cattle egrets (*Bubulcus ibis*) follow tractors to feed on invertebrates unearthed during ploughing. A similar behavior has been described for certain granivorous birds that take unearthed seeds (e.g., Whittingham et al. 2006). Finally, certain species may be favored by the lower vegetation cover and the consequent higher antipredator visibility in ploughed fields (Butler et al. 2005), although this possibility has not been tested in this study.

During the mating season, habitat and food models basically coincided. The retained variables were surfaces of various substrates, but no food biomass estimates were included as predictors. This result contrasts with some previous studies (Traba et al. 2008; Concepción and Díaz 2010) which suggest that food

availability is a key factor during the mating season. We can think of two possible reasons for the absence of a significant effect of biomass variables during this period of the annual cycle. First, during the mating season, weeds and invertebrates are more abundant in our non-intensive farmland area compared with other periods of the year. Food availability may then be higher than demands and thus represent no limiting factor for bird abundance and diversity. Consequently, the effect of food biomass in each particular substrate type may be obscured during this season. Second, during the mating season, most birds are involved in defending territories, pairing and searching for nest sites. For these tasks, a rough estimate of available surfaces of the different field types may be a better indicator of habitat suitability than a precise estimate of food availability. Substrate selection during this season may indeed provide enough information about the best place to nest and the food availability for chicks expected later in the season. Third, bird abundance, diversity, and richness in spring are limited by other variables such as territoriality, and complex intra and interspecific interactions within the breeding bird community. In spite of the absence of a clear effect of food biomass during the mating season, the surface of legumes was correlated with the biomass of arthropods in this substrate (0.81), showing its role in increasing food availability. Previous studies had already described the importance of legumes as a source of nitrogen for many species (Karasov 1990), and in particular for steppe birds in dry farmland areas (Bretagnolle et al. 2011; Bravo et al. 2012). Our study suggests that legumes may also be important as a food source for insectivorous species.

Unlike during winter, AES cereal stubbles were the most important predictor in spring averaged model for bird abundance. For birds, natural and AES-managed cereal stubbles are probably identical during most part of the winter, but on managed stubbles, AES restrictions prevent herbicide and insecticide use between harvest and ploughing. This fact surely favored the growth of abundant weeds on managed stubbles in spring and released the observed positive response from birds to the AES stubbles during the mating season. According to the AES rules, ploughing is allowed in January–March, but later forbidden until July. This was an unexpected positive effect of the AES, as we thought all farmers would plough AES stubbles before the deadline of 31 March, as they did with non-AES

stubbles (which are also sprayed usually with chemical products against weeds). It is necessary to highlight that AES stubbles were kept from spring to the following autumn without any additional payments to farmers.

During the postfledging season, habitat and food models were also different. While in habitat models, the surface of fallows and ploughed fields with abundant sprouted weeds were the main predictors, food models showed that the birds' response was really induced by the biomass of arthropods in non-AES fallows. In food models, the surface of ploughed fields with sprouted weeds is still retained as a significant predictor. This is because in dry cereal farmland areas ploughed fields with sprouted weeds are the only substrate where birds can find green plants and associated canopy arthropods in summer.

CONCLUSIONS

Our study showed that the AES contributed to increase the abundance and diversity of farmland birds in our study area. The positive responses observed in four variables analyzed were in part induced by some of the AES measures applied. However, many land-use variables not regulated by the AES were also important, probably due to the extensive agricultural regime predominant in our study area. The models presented in our study enabled evaluating the percent benefit obtained from AES measures as compared to non-AES land-use variables.

Exhaustive field work was devoted in this study to measure landscape complexity, vegetation structure and food availability, all of which are considered important factors influencing bird behavior and distribution patterns. In our study, various important effects of seed and arthropod biomass detected using food models would have gone unnoticed using habitat models where only substrate composition is measured. This was at the cost of a much higher investment in time and personnel in food models, with a consequent increase in the total cost of the research. However, detailed food measurements allowed increasing the explanatory power of models describing bird responses, as well as identifying the causes underlying these responses.

Our study finally highlighted the need to apply AES measures and to study bird responses separately in different periods of the year. As most birds use different substrates throughout the annual cycle, the same AES measures may have different effects in different seasons. Thus, farmland areas need to be managed from that seasonal perspective to maximize the benefits of AES programs. In our case, the AES measures aimed at enhancing the benefits of traditional farming cycles at dry cereal areas, by providing supplementary legume crops and fallows and limiting tillage work.

ACKNOWLEDGEMENTS

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Appendix S1. List of bird species (in alphabetical order) contacted during surveys of the study area, indicating their total abundance (sum of abundance values in all transects) in each season and the Species of European Conservation Concern (SPEC) category.

Species	SPEC	Wintering	Mating	Post-fledging
Accipiter gentilis	*NON-SPEC*			1
Alauda arvensis	*SPEC 3*	8625	6	5
Alectoris rufa	*SPEC 2*	185	108	96
Anas platyrhynchos	*NON-SPEC*		21	1
Anthus campestris	*SPEC 3*		14	
Anthus pratensis	*NON-SPEC*	246		
Apus apus	*NON-SPEC*		2	9
Aquila adalberti	*SPEC 1*		1	
Asio flammeus	*SPEC 3*	7		
Athene noctua	*SPEC 3*			3
Bubo bubo	*SPEC 3*		1	
Bubulcus ibis	*NON-SPEC*	29	16	185
Burhinus oedicnemus	*SPEC 3*		66	110
Buteo buteo	*NON-SPEC*	10	3	5
Calandrella brachydactila	*SPEC 3*		72	
Carduelis cannabina	*SPEC 2*	967	110	77
Carduelis carduelis	*NON-SPEC*	362	75	121
Carduelis chloris	*NON-SPEC*	99	43	11
Ciconia ciconia	*SPEC 2*		4	16
Circaetus gallicus	*SPEC 3*			5
Circus aeruginosus	*NON-SPEC*	7	9	6
Circus cyaneus	*SPEC 3*	8	6	3
Circus pygargus	*NON-SPEC*		21	19
Cisticola juncidis	*NON-SPEC*	5	74	52
Clamator glandarius	*NON-SPEC*	3	17	2
Columba livia	*NON-SPEC*	120	98	461
Columba palumbus	*NON-SPEC*	33	84	395
Corvus corax	*NON-SPEC*		3	1
Corvus corone	*NON-SPEC*	4		7
Corvus monedula	*NON-SPEC*	29	25	14
Coturnix coturnix	*SPEC 3*	3	102	16
Delichon urbicum	*SPEC 3*		46	45
Emberiza schoeniclus	*NON-SPEC*	4		
Falco columbarius	*NON-SPEC*	7		
Falco naumanni	*SPEC 1*		11	
Falco peregrinus	*NON-SPEC*			1
Falco subbuteo	*NON-SPEC*			3
Falco tinnunculus	*SPEC 3*	10	9	24
Fringilla coelebs	*NON-SPEC*	84	6	
Galerida cristata	*SPEC 3*	139	265	336
Gelochelidon nilotica	*SPEC 3*			10
Hieraaetus pennatus	*SPEC 3*		1	
Hippolais polyglotta	*NON-SPEC*		1	
Hirundo daurica	*NON-SPEC*		1	
Hirundo rustica	*SPEC 3*		81	43
Lanius meridionalis	*SPEC 3*	12	4	12
Lanius senator	*SPEC 2*		3	16
Luscinia megarhynchos	*NON-SPEC*		1	

Species	SPEC	Wintering	Mating	Post-fledging
Melanocorypha calandra	*SPEC 3*	524	660	712
Merops apiaster	*SPEC 3*		7	16
Miliaria calandra	*SPEC 2*	648	564	117
Milvus migrans	*SPEC 3*		5	14
Milvus milvus	*SPEC 2*	4		
Motacilla alba	*NON-SPEC*	98		
Motacilla cinerea	*NON-SPEC*		11	
Motacilla flava	*NON-SPEC*		30	
Oenanthe hispanica	*SPEC 2*		78	64
Oenanthe oenanthe	*SPEC 3*		48	31
Otis tarda	*SPEC 1*	541	280	160
Parus caeruleus	*NON-SPEC*	1		
Parus major	*NON-SPEC*			1
Passer domesticus	*SPEC 3*	305	118	165
Passer hispaniolensis	*NON-SPEC*	80	52	139
Passer montanus	*SPEC 3*	142	82	61
Petronia petronia	*NON-SPEC*	56	42	43
Phoenicurus ochruros	*NON-SPEC*	12		
Phoenicurus phoenicurus	*SPEC 2*		1	
Phylloscopus collybita	*NON-SPEC*	4	1	
Pica pica	*NON-SPEC*	169	192	241
Picus viridis	*SPEC 2*	1		1
Pluvialis apricaria	*NON-SPEC*	41		
Pterocles alchata	*SPEC 3*	12	2	9
Pterocles orientalis	*SPEC 3*	84	28	7
Saxicola rubetra	*NON-SPEC*		1	
Saxicola torquata	*NON-SPEC*	6	2	14
Serinus serinus	*NON-SPEC*	110	98	95
Sturnus unicolor	*NON-SPEC*	740	186	1086
Sylvia cantillans	*NON-SPEC*		1	3
Sylvia melanocephala	*NON-SPEC*	6	3	3
Tetrax tetrax	*SPEC 1*	225	140	63
Tringa ochropus	*NON-SPEC*			1
Turdus merula	*NON-SPEC*	3		
Turdus philomelos	*NON-SPEC*	31		
Upupa epops	*SPEC 3*	1	9	16
Vanellus vanellus	*SPEC 3*	368	15	1
Total birds		15210	4066	5174

Appendix S2. Correlation values between consecutive transects for each season. Autocorrelation coefficient values (ρ) calculated between each pair of consecutive transects (lag=1) are shown. Asterisks show 0.05>P-values>0.01. There was no autocorrelation coefficient with a P-value <0.01. In each season 64 autocorrelation coefficients were calculated. Because this is a large number of tests, it was expected that a few of them would be significant by chance. Therefore, we applied the Bonferroni correction to the P-values (Bonferroni-corrected P values were 0.05/64=0.00078) and observed no significant autocorrelation values after correction.

1) Wintering season. Four (6.3%) out of 64 coefficients (marked with *) were significant before Bonferroni correction

Site	Abundance				Richness				Diversity				Spec-score			
	2006	2007	2008	2009	2006	2007	2008	2009	2006	2007	2008	2009	2006	2007	2008	2009
1	0.16	-0.52	-0.55	0.26	0.60*	-0.07	-0.56	0.31	0.15	-0.10	-0.16	0.30	0.55	-0.70*	0.03	0.21
2	0.02	0.43	-0.07	0.05	0.01	-0.06	-0.19	0.19	0.03	0.05	-0.21	-0.21	-0.05	0.11	-0.02	0.05
3	0.13	-0.51	-0.26	-0.27	0.64*	-0.06	0.23	-0.06	-0.20	0.20	0.31	0.20	0.26	-0.78*	0.15	-0.30
4	-0.36	0.10	-0.04	-0.05	0.18	-0.03	-0.15	-0.03	-0.19	-0.16	0.21	0.23	-0.35	0.09	-0.05	-0.06

2) Mating season. Two (3.1%) out of 64 coefficients (marked with *) were significant before Bonferroni correction

Site	Abundance				Richness				Diversity				SPEC-score			
	2006	2007	2008	2009	2006	2007	2008	2009	2006	2007	2008	2009	2006	2007	2008	2009
1	-0.42	0.35	0.24	0.01	-0.33	0.63*	0.00	-0.05	-0.22	0.24	-0.11	-0.08	-0.02	0.51	0.34	0.48
2	0.42	0.10	-0.85*	-0.08	0.31	-0.51	-0.02	-0.18	0.31	-0.59	0.20	-0.11	-0.30	-0.58	0.36	0.25
3	-0.50	0.07	-0.57	-0.33	-0.66	0.41	-0.02	0.06	-0.50	0.25	0.05	0.43	-0.17	0.03	-0.45	-0.32
4	-0.50	-0.37	-0.28	-0.06	0.00	0.18	0.13	-0.31	0.10	0.22	0.15	-0.31	-0.50	-0.37	-0.28	-0.06

3) Post-fledging season. One (1.6%) out of 64 coefficients (marked with *) were significant before Bonferroni correction

	Abundance				Richness				Diversity				SPEC-score			
Site	2006	2007	2008	2009	2006	2007	2008	2009	2006	2007	2008	2009	2006	2007	2008	2009
1	0.14	-0.03	0.63*	-0.53	-0.29	0.59	0.15	0.18	-0.38	-0.23	0.15	-0.23	-0.48	0.41	-0.24	-0.53
2	-0.19	-0.44	-0.14	-0.10	-0.15	-0.13	-0.22	0.22	0.14	0.37	0.06	0.33	0.13	0.08	-0.09	-0.10
3	-0.12	-0.17	-0.21	-0.40	-0.29	0.22	-0.18	0.22	-0.22	-0.13	-0.01	-0.13	0.03	0.01	-0.29	-0.40
4	-0.57	-0.26	0.35	-0.23	-0.35	-0.23	0.40	0.33	-0.23	-0.37	0.52	0.48	-0.52	-0.26	0.23	-0.23

Appendix S3. Variables defined to measure habitat characteristics and biomass of arthropods and seeds, indicating the model type (food o habitat) in which they were included. Surface variables were measured as percentages per transect, biomass variables as grams per transect, and height as centimetres per transect. All vegetation and ground structure variables calculated for each transect were derived from the mean value in each field type and site and according to its surface.

Short name	Definition	Model
ArthrAES	Total arthropod biomass in the sum of all AES fallows,+ AES cereal stubbles + AES legume fields + AES legume stubble fields	Food
ArthrCerNAT	Arthropod biomass in non-AES cereal fields	Food
ArthrCerStubbleAES	Arthropod biomass in AES cereal stubble fields	Food
ArthrCerStubbleNAT	Arthropod biomass in non-AES cereal stubble fields	Food
ArthrEdge	Arthropod biomass in edges	Food
AES	Surface of AES	Habitat
ArthrFallowAES	Arthropod biomass in AES fallow fields	Food
ArthrFallowNAT	Arthropod biomass in non-AES fallow fields	Food
ArthrHQFNAT	Arthropod biomass in the sum of all non-AES 'high quality fields' (non-AES fallows + non-AES cereal stubble + non-AES legumes + non-AES legume stubbles)	Food
ArthrLegAES	Arthropod biomass in AES legume fields	Food
ArthrLegNAT	Arthropod biomass in non-AES legume fields	Food
ArthrLegStubbleAES	Arthropod biomass in AES legume stubble fields	Food
ArthrLegStubbleNAT	Arthropod biomass in non-AES legume stubble fields	Food
ArthrNAT	Arthropod biomass in non-AES fields	Food
ArthrPlough	Arthropod biomass in ploughed fields	Food
ArthrPlough2	Arthropod biomass in ploughed fields with sprouted weeds	Food
ArthrTot	Total arthropod biomass	Food
CerNAT	Surface of cereal fields	Habitat
CerStubbleAES	Surface of cereal stubble fields included in AES	Habitat
CerStubbleNAT	Surface of cereal stubble fields not included in AES	Habitat
CerStubbleTot	Total surface of cereal stubble	Habitat
CoverAES	Mean vegetation cover derived from AES in percentage	Food
CoverTOTAL	Mean vegetation cover	Food
DifHeight	Difference between the maximum and minimum vegetation height in cm	Food
Edge	Surface of edges	Habitat
FallowAES	Surface of fallow fields included in AES	Habitat
FallowNAT	Surface of fallow fields not included in AES	Habitat
FallowTot	Total surface of fallow fields	Habitat
HQFNAT	Surface of non-AES 'high quality fields' (non-AES fallows + non-AES cereal stubble + non-AES legumes + non-AES legume stubbles)	Habitat
LandscapeDiversity	Landscape diversity (Shannon index)	Habitat
LandscapeDiversityAES	Landscape diversity (Shannon index) associated with AES	Habitat
LandscapeDiversityNAT	Landscape diversity (Shannon index) not associated with	Habitat

Short name	Definition	Model
	AES	
Leg	Total surface of legume fields	Habitat
LegAES	Surface of legume fields included in AES	Habitat
LegNAT	Surface of legume fields not included in AES	Habitat
LegStubbleAES	Surface of legume stubble fields included in AES	Habitat
LegStubbleNAT	Surface of legume stubble fields not included in AES	Habitat
LegStubbleTot	Total surface of legume stubble fields	Habitat
MeanHeight	Mean vegetation height in cm	Food
Plough	Surface of ploughed fields	Habitat
Plough2	Surface of ploughed fields with sprouted weeds	Habitat
Roughness	Roughness of the ground using three categories (low, medium or high)	Food
SeedAES	Seed biomass in AES fallows + AES cereal stubble + AES legume + AES legume stubble	Food
SeedCerNAT	Seed biomass in non-AES cereal fields	Food
SeedCerStubbleAES	Seed biomass in AES cereal stubble fields	Food
SeedCerStubbleNAT	Seed biomass in non-AES cereal stubble fields	Food
SeedEdge	Seed biomass in edges	Food
SeedFallowAES	Seed biomass in AES fallow fields	Food
SeedFallowNAT	Seed biomass in non-AES fallow fields	Food
SeedHQFNAT	Seed biomass in non-AES 'high quality fields' (non-AES fallows + non-AES cereal stubble + non-AES legumes + non-AES legume stubbles)	Food
SeedLegAES	Seed biomass in AES legume fields in AES	Food
SeedLegNAT	Seed biomass in non-AES legume fields	Food
SeedLegStubbleAES	Seed biomass in AES legume stubble fields	Food
SeedLegStubbleNAT	Seed biomass in non-AES legume stubble fields	Food
SeedNAT	Seed biomass in non-AES fields	Food
SeedPlough	Seed biomass in ploughed fields	Food
SeedPlough2	Seed biomass in ploughed fields with sprouted weeds	Food
SeedTot	Total seed biomass	Food

Appendix S4. Results of the Principal Component Analyses (PCA) carried out to explore correlations among variables. The most representative variable from each axis (marked in bold) was selected to be included it in the candidate models for bird abundance, richness, diversity and SPEC-score. The type of model (food, habitat) to which each variable belongs is shown for the first set of models.

1) Food models

Wintering	Variable		PC 1	PC 2	PC 3	PC 4	PC 5	PC 6	PC 7	PC 8	PC 9	PC 10	PC 11
	ArthrLegStubbleNAT	Food	0.018	0.006	-0.029	0.004	-0.039	0.021	0.051	0.047	0.081	**-0.968**	-0.003
	ArthrTot	Food	0.041	**-0.954**	0.014	-0.066	0.011	-0.063	-0.022	0.003	-0.069	0.015	0.001
	CerStubbleNAT	Habitat	-0.034	0.0314	0.057	**-0.933**	0.005	0.042	-0.049	-0.121	0.068	0.010	0.061
	FallowAES	Habitat	**0.871**	0.021	-0.061	0.029	-0.08	-0.016	0.142	-0.066	0.167	-0.132	0.049
	LegNAT	Habitat	-0.063	-0.031	-0.019	0.042	0.023	0.019	0.030	-0.051	0.024	-0.121	**-0.958**
	LegStubbleAES	Habitat	0.058	0.004	-0.007	0.016	**-0.928**	0.031	-0.026	0.004	0.001	-0.065	-0.003
	Plough	Habitat	-0.091	-0.002	-0.132	0.231	0.046	0.122	0.158	0.066	**-0.851**	0.039	0.091
	SeedCer	Food	0.033	0.153	0.229	0.084	0.052	0.081	-0.020	**-0.768**	0.084	0.02	0.144
	SeedHQFNAT	Food	0.052	-0.129	**0.932**	-0.102	0.029	0.007	0.110	-0.002	0.099	-0.006	-0.065
	SeedLegAES	Food	0.055	-0.025	-0.091	0.042	-0.008	**-0.937**	0.056	-0.031	0.054	0.032	0.066

Mating	Variable	Model	PC 1	PC 2	PC 3	PC 4	PC 5	PC 6	PC 7	PC 8
	ArthrNAT	Food	0.086	-0.174	0.014	0.486	-0.021	0.084	**0.835**	-0.061
	CerStubbleAES	Habitat	-0.166	0.015	0.003	-0.064	**-0.951**	-0.066	-0.036	0.021
	FallowNAT	Habitat	0.033	-0.249	0.012	**0.931**	0.085	-0.055	-0.022	-0.034
	DifHeight	Food	-0.115	0.001	**-0.936**	-0.063	0.145	-0.069	0.018	0.031
	LegAES	Habitat	**-0.942**	0.034	-0.069	0.015	0.028	-0.031	-0.122	-0.026
	LegNAT	Habitat	0.068	0.081	-0.049	0.138	0.103	0.081	-0.016	**-0.917**
	Plough2	Habitat	0.165	-0.121	0.006	-0.068	0.057	**0.908**	-0.006	0.009
	SeedFallowNAT	Food	0.041	**-0.947**	-0.011	0.280	0.021	0.038	0.016	0.012

Post-fledging	Variable	Model	PC 1	PC 2	PC 3	PC 4	PC 5	PC 6	PC 7	PC 8	PC 9
	ArthrFallowNAT	Food	-0.043	**0.921**	-0.049	0.018	0.071	0.01	0.003	0.001	-0.006
	ArthrLegNAT	Food	-0.023	-0.02	0.022	0.021	**-0.995**	0.004	-0.009	-0.03	0.024
	CerStubbleAES	Habitat	0.288	0.016	0.006	0.072	0.01	0.116	-0.052	**0.941**	0.001
	FallowAES	Habitat	**0.947**	-0.033	0.016	0.063	0.021	-0.037	0.051	0.23	-0.019
	LegStubbleAES	Habitat	0.261	-0.016	-0.008	-0.103	0.01	**-0.972**	-0.051	0.001	0.028
	LegStubbleNAT	Habitat	-0.041	0.065	0.021	**-0.952**	0.01	0.034	0.023	-0.057	0.014
	Plough	Habitat	-0.159	-0.314	**-0.912**	0.098	0.004	0.006	0.152	-0.089	0.086
	Plough2	Habitat	0.036	-0.048	-0.051	0.032	0.041	0.066	0.007	-0.003	**-0.943**
	SeedsLegAES	Food	0.169	-0.058	-0.061	0.036	0.039	-0.089	**-0.957**	0.2162	-0.001

2) Habitat models

Wintering	Variable	PC 1	PC 2	PC 3	PC 4	PC 5	PC 6	PC 7	PC 8	PC 9	PC 10	PC 11
	CerStubbleNAT	-0.031	0.341	0.036	**-0.933**	0.007	0.041	-0.051	-0.112	0.059	0.002	0.061
	Edge	-0.020	0.018	-0.015	0.061	0.03	-0.044	0.325	**0.726**	-0.086	-0.043	0.213
	FallowAES	**0.872**	0.014	-0.046	0.039	-0.052	-0.015	0.21	-0.074	0.201	-0.141	0.061
	FallowNAT	-0.05	**-0.723**	0.312	0.165	-0.031	-0.052	0.468	0.219	0.044	0.036	-0.019
	LegAES	0.351	0.01	0.102	0.002	0.051	**-0.831**	-0.069	0.171	0.051	0.029	-0.009
	LegNAT	-0.062	-0.043	-0.036	0.026	0.019	0.019	0.034	-0.036	0.051	-0.113	**-0.972**
	LegStubbleAES	0.054	0.009	-0.008	0.021	**-0.934**	0.061	-0.022	0.003	0.002	-0.084	-0.002
	LegStubbleNAT	0.047	0.012	-0.044	0.026	-0.128	0.039	0.05	0.047	0.071	**-0.964**	-0.005
	Plough	-0.122	-0.005	-0.121	0.32	0.041	0.144	0.155	0.057	**-0.851**	0.062	0.072
	SeedFallowNAT	0.058	-0.234	**0.876**	-0.01	0.003	0.012	0.132	0.119	0.064	0.029	0.011

Mating	Variable	PC 1	PC 2	PC 3	PC 4	PC 5	PC 6	PC 7	PC 8
	ArthrCerNAT	0.21	0.121	-0.232	-0.253	-0.068	0.018	**0.731**	-0.056
	CerStubbleAES	-0.161	0.015	0.002	-0.063	**-0.941**	-0.052	-0.032	0.048
	FallowNAT	0.034	-0.221	0.012	**0.931**	0.08	-0.05	-0.019	-0.07
	LegAES	**-0.924**	0.014	-0.061	0.011	0.019	-0.024	-0.113	-0.012
	LegNAT	0.071	0.049	-0.041	0.131	0.051	0.082	-0.032	**-0.916**
	Plough	0.037	0.18	**0.861**	-0.166	0.135	-0.158	-0.12	0.063
	Plough2	0.151	-0.116	0.04	-0.04	0.062	**0.921**	-0.007	0.008
	SeedFallowNAT	0.026	**-0.947**	-0.009	0.2	0.018	0.072	0.041	0.03

Post-fledging	Variable	PC 1	PC 2	PC 3	PC 4	PC 5	PC 6	PC 7	PC 8	PC 9
	CerStubbleAES	0.158	0.003	0.004	0.027	0.01	0.018	-0.066	**0.928**	0.01
	FallowAES	**0.961**	-0.017	0.012	0.051	0.008	-0.035	0.079	0.26	-0.039
	FallowNAT	-0.05	**0.847**	-0.069	-0.258	0.081	-0.034	0.069	0.002	-0.019
	LegAES	0.406	-0.026	-0.059	0.047	0.061	-0.269	**-0.915**	0.121	0.011
	LegNAT	-0.069	-0.038	0.051	0.029	**-0.974**	0.013	-0.005	-0.061	0.066
	LegStubbleAES	0.174	-0.041	-0.012	-0.019	0.006	**-0.963**	-0.042	0.00	0.039
	LegStubbleNAT	-0.072	0.109	0.009	**-0.957**	0.021	0.054	0.044	-0.019	0.013
	Plough	-0.151	-0.301	**-0.897**	0.151	0.011	0.007	0.133	-0.047	0.156
	Plough2	0.027	-0.045	-0.061	0.049	0.033	0.081	0.013	-0.002	**-0.927**

162

CAPÍTULO 5

Este capítulo se encuentra en fase de preparación:

Ponce, C., Salgado, I., Bravo, C., Gutiérrez, N. & Alonso, J.C. Effects of farming practices on nesting success of steppe birds in dry cereal farmland.

CAPÍTULO 5

Effects of farming practices on nesting success of steppe birds in dry cereal farmland

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ABSTRACT

Predation is the most common cause of nest failure in ground-nesting birds. Natural predation rates may be influenced by both, regular agricultural practices and agri-environmental measures promoted by agri-environmental schemes (AES). These practices and measures could increase predation rates by increasing predator abundance, or decrease them by providing more vegetation where birds should find safe nesting places. We investigated these issues in a dry cereal farmland in central Spain, by means of an experimental setup with 520 artificial nests of quail. Artificial nests were distributed among 13 sites, each including all main field types of the area. Predation rate was analyzed using averaged mixed models, and predictor variables describing the physical characteristics of the nesting site at three scales (landscape, field, and nest location within each field - central and peripheral-). Game cameras were used to identify predators and analyze their nest predation patterns. We found that 6.2% of the nests were destroyed by tractors ploughing in spring. Overall nest predation rate was 66.3%, affecting more to nests surrounded by organic cereal crops and ploughed fields. Nests located near field margins suffered more predation than those in the centre of the fields. Nest predation was highest in ploughed fields, intermediate in AES-promoted fields (fallows, organic cereal, vetch), and lowest in regular cereal fields. In all field types nest predation rate decreased with increasing vegetation height, because tall vegetation offered good nest concealment opportunities. Fallows, vetch fields and organic cereal fields provided intermediate vegetation heights, and thus relatively safe nesting sites. However, but due to the high food availability predators could find on these substrates, they acted as ecological traps because predators concentrated their predation activity on them. Camera traps recorded 42 predation events (81% of birds, 19% of mammals). The main predators were Marsh Harrier, Montagu's Harrier and Common Buzzard. Spring ploughing should be restricted to prevent nest destruction. Fields promoted by AES should be dispersed in order to avoid attracting nest predators.

Keywords

INTRODUCTION

Dry cereal farmland holds important breeding and wintering populations of many farmland bird species of European conservation concern (Butler et al. 2010). Although agriculture has favoured the expansion of farmland habitats in past centuries, the replacement of traditional by intensive farming practices in recent times has led to habitat changes whose negative effects have been highlighted in numerous studies. The main practices associated to intensive agriculture are land management changes (e.g., moving the ploughing of cereal stubbles forward by several months, from spring to immediately after harvest), loss of crop diversity, increased pesticide and fertilizer use, removal of edges and uncultivated areas, and earlier harvest dates (Newton 2004). All these changes have resulted in the loss of suitable feeding and nesting habitats, and a reduction in food and nesting places available. One of the main consequences of this habitat deterioration has been a significant decline suffered by European farmland bird populations during the last decades (Donald *et al.* 2001, Gregory *et al.* 2005). Like most bird populations in northern and central Europe, those of the Mediterranean region have also suffered the consequences of recent changes in farmland habitats. For example, according to the last Common Breeding Bird Monitoring Scheme report in Spain (SACRE), the numbers of farmland breeding birds have declined by 25% during the last 17 years, and almost 30% inside Important Bird Areas (Escandell 2015).

Predation has been identified as the most important cause of nest failure of ground-nesting birds in farmland habitats (Draycott *et al.* 2008), and thus a relevant factor determining their decline (Donald *et al.* 2002, Bradbury *et al.* 2000). High predation rates may limit the breeding populations of farmland species (Gibbons *et al.* 2007), or influence their demographic parameters (Whittingham & Evans 2004). Thus, an interesting research issue is how intensive agricultural practices interact with nest predation. It is known that intensive agriculture leads to increasing predation rates in farmland species (Tapper *et al.* 1996, Paridis *et al.* 2000, Newton 2004). At least three hypotheses have been proposed to explain this phenomenon. The first is that high predation rates could

be the result of a higher density of predators (Baillie *et al*. 2002, 2007, Evans 2004). A second hypothesis suggests that predation rates can increase independently of changes in predator density (Wilson *et al*. 1997a, Donald 1999, Pescador & Peris 2001). For example, a decrease in suitable habitat availability can make prey more vulnerable to predation (Whittingham & Evans 2004) or force some species to concentrate their nests in the remaining smaller patches of appropriate habitat, which would suffer higher predation pressure and thus turn into "*ecological traps*" (Chamberlain *et al*. 1995, Pescador & Peris 2001). Also, predators can shift their diet and select new prey if their usual target species have declined as a consequence of agricultural intensification (Schmidt 1999, Evans 2004, Newton 2004,). In fact, some of the characteristics of agricultural intensification are the destruction of edges and fallows, or the homogenization of crops, leaving only small isles of suitable habitat for nesting. A third hypothesis suggests that if removing edges and fallows forces ground-nesting species to nest in more exposed places, nest predation will increase. Whatever the reason, the link between increasing nest predation rates and agricultural intensification seems clear.To reverse the decline of farmland birds, agri-environmental schemes (AES) have been running in many countries over the last decades (reviewed in Kleijn & Sutherland 2003). Most AES include payments to farmers for implementing measures that benefit wildlife (Kleijn & Sutherland 2003). While studies evaluating and proposing actions intended to increase food availability are relatively abundant (Campbell *et al*. 1997, Herkert 2009, Lapiedra *et al*. 2011), those relating nesting habitat quality and predation rates are less common (but see Beja *et al*. 2013, Evans 2004, Fletcher *et al*. 2010).

Increasing natural vegetation (e.g., by creating fallows) and landscape heterogeneity (e.g., by introducing different crops) are some of the measures usually proposed in AES (Kleijn & Sutherland 2003). Besides providing food resources, these measures contribute to reduce nest predation rates by providing safe nesting places (Newton 1998, Wilson *et al*. 2001a, Beja *et al*. 2013), or by increasing the number of potential nest sites that predators must search, thus reducing nest density in each field and improving nest concealment (Bowman & Hams 1980, Martin 1993). But increasing high quality habitats may also attract predators (Pescador & Peris 2001).

Although nest concealment is an important factor influencing predation rate on farmland habitats (Yanes & Suárez 1995, Magaña *et al.* 2010), the type of predator is also relevant (Patten & Bolger 2003. Bayne *et al.* 1997). Therefore, understanding how the foraging patterns of all potential predators relate to nesting sites, landscape and habitat characteristics is crucial for implementing appropriate management actions (Benson *et al.* 2010). Also, birds use visual stimuli for nest detection, whereas mammals use the sense of smell and thus, nest concealment should not be a crucial factor in mammal nest predation avoidance (Rangen et al. 1999). Studying the predator community seems therefore essential to understand the reasons of the success or failure of AES involving nest concealment.

In this study we assessed nest predation in relation to both, ordinary agricultural practices applied in Mediterranean dry cereal farmland, as well as additional measures from a currently running AES, in a farmland area of central Spain. We used an experimental design with artificial nests, and game cameras to detect and identify predators. We also considered the influence of nest location within the field and the characteristics of the surrounding habitat (landscape context), since these factors can modify nest predation (Storch *et al.* 2005, Reino *et al.* 2010). Our hypothesis was that nest predation would be affected by the micro-habitat structure around nests, the landscape characteristics, and the differences in foraging patterns among predators. Specifically, we predicted that increasing vegetation height and cover, and landscape diversity derived from AES measures would reduce predation rates by favouring nest concealment and providing suitable nesting places. We also predicted that predation rates would be higher on edges, which are commonly used by some predators (Blouin-Demers & Weatherhead 2001). In light of the results, some management actions that could contribute to reduce nest predation are discussed.

METHODS

Study area, field selection and nest placement

The study was carried out in the Special Protected Area (SPA) 139 "*Estepas cerealistas de los ríos Jarama y Henares*", located in the north-eastern part of Madrid province (central Spain), where an agri-environmental scheme (AES) has been running since 2003. Briefly, the AES consisted on payments to farmers for growing organically vetch and cereal, interruption of the cereal production (long-term fallows) and maintenance of cereal stubble during the winter (for more details on AES measures see Ponce *et al.* 2014). The landscape in the SPA is homogeneous, and the area is mainly dedicated to dry cereal cultivation (wheat *Triticum aestivum*, barley *Hordeum vulgare*, and oat *Avena sativa*) with some dispersed bushes (*Retama sphaerocarpa*, *Thymus spp.*, etc.) and sporadic trees (*Quercus ilex*, *Pinus spp.*). Some of the most common ground-nesting birds in the SPA are Calandra Lark (*Melanocorypha calandra*), Corn Bunting (*Miliaria calandra*), Great Bustard (*Otis tarda*), Crested Lark (*Galerida cristata*) and Little Bustard (*Tetrax tetrax*) (Ponce *et al.* 2014), all of them included in the list of Species of European Conservation Concern (BirdLife International 2004).

There is no consensus regarding the applicability of predation rates obtained in experimental studies with artificial nests to natural situations (Martin 1987 and Matessi & Bogliani 1999, Mezquida & Marone 2003, Robinson *et al.* 2005, Noske *et al.* 2008). However, artificial nests are considered useful for identifying factors affecting spatial and temporal predation patterns (Major & Kendall 1996, Roos 2002, Batary & Baldi 2005, Ludwig *et al.* 2012, Mandema *et al.* 2013) in comparative studies and different settings. Also, nest predation of artificial nests seems to be useful for establishing relative predation pressures, at least for ground-nesting passerines in open habitats with low structural complexity, where rates from experimental and natural situations were found to be similar (Vögeli *et al.* 2011). Furthermore, artificial nests allow using sample sizes in habitats that may be avoided by birds and controlling the parameters to be studied (Beja *et al.* 2013).

To place artificial nests we selected 13 sites in the SPA (Figure 1). Each of these 13 sites included all kind of fields from the AES (long-term fallows, vetch and organically cultivated cereal) and those from regular agricultural activities (ploughed fields and cereal crops). We were particularly interested in investigating the effects of fields promoted by AES (fallows, vetch and organic cereal) on nest predation rate. We selected fields as close together as possible to maximize the probability that all fields of that site were located within the territory of a given predator, and thus the predation probability was the same among fields within a given site.

Figure 1. Location of the study area in the Iberian Peninsula. The figure on the right shows the Special Protected Area (SPA) 139 "*Estepas cerealistas de los ríos Jarama y Henares*" and the 13 sites where the field work was carried out.

We carried out the experiment in two trials during the spring in 2012. Artificial nests were placed on 16th May for the first trial and on 5th June for the second trial. We monitored nests during 14 days which is equivalent to a nesting cycle of most passerine species present in the study area. We removed non-predated nests at the end of each trial. We also left 1 week between the end of the first trial and the start of the next one, to avoid any habituation of predators. We placed 4 artificial nests in each field totalling 520 nests (260 for each trial, and 20 nests at each site). Within a field, we placed 2 nests close to the edge (at 2 m) and 2

in well inside the field (at 50 m from the edge nest towards the centre of the field; this distance is higher than those considered in most studies, see e.g. Díaz & Carrascal 2006). This design minimizes the probability that two nests of the same field are found by the same predator, and thus maximizes independence of predation results among nests. Each artificial nest consisted of a slight depression on the ground to avoid their displacement, with 3 fresh Quail (*Coturnix sp.*) eggs. We used eggs bought at industrial quail farms because they are easy to obtain in large quantities and similar to those that can be found naturally in farmland areas (de Graaf & Maier 1996, Maier & de Graaf 2001). We used eggs from two different companies, but showing no differences in length, width and weight (n=150, p >0.43 in all cases). Close to each nest we placed a reed (1.20 m height) to enable an easy location of the nest in subsequent visits. A small tag with information about the study was attached to the top end of the reed.

Since predators were identified by means of game cameras (see below), we also tested the possible effect of these cameras on nests predation (Richardson *et al*. 2009). We monitored 41 nests with game cameras and compared them with a sample of 46 control nests without game cameras. All of them were visited 3 times (to avoid the "researcher effect") evenly distributed along the study periods, and after cameras were installed. We did not find any effect of placing game cameras on artificial nests predation (Chi-Square = 0.01, P= 0.93). Since some studies have shown that increasing the number of visits has an effect on nest predation (e.g., Verboven *et al*. 2001, but see Ibáñez-Álamo *et al*. 2012) we also tested the researcher effect. We visited 46 nests 3 times and 46 only 1 (plus the placement day in both samples). We did not find any effect of the number of visits (Chi-Square = 0.21, P= 0.65). Accordingly, all artificial nests were included in subsequent analyses.

Field and landscape variables description

We distinguished three groups of variables: those related to the micro-habitat structure around the nest (measured within a circle of 2 m diameter centred on the nest); those related to the nest location, and those describing the landscape context, measured within a circle of 100 m radius around the nest, since it is known that landscape composition and structure may affect predator composition

and abundance (Pita *et al.* 2009). Prior to starting the nest visits, all observers participating in the fieldwork spent one day standardizing the measures and estimates to be taken.

For each nest we recorded the UTM coordinates using a GPS (Garmin, ±2 m error). Vegetation abundance and structure were assessed visually around the nest. Data recorded were total plant cover (% of the surface) and vegetation height (maximum, mean and most frequent, in cm). Nest location variables were the field type (ploughed, cereal crop, organic cereal crop, vetch and fallow) and the location in the field (edge or interior). Landscape variables were distances (in m) to the nearest shrub or tree (estimated visually), and watercourses and paths, presence of a watercourse, length of watercourses and edges (in m), and the surface of each type of field plus the surface of buildings (including farms) and shrubs (based on GIS information).

Video monitoring

We installed 21 game cameras (model Bresser 3310000) at random nests trying that they were evenly distributed among sites, field types, and location in the field (edge, interior). The aim was to record any predation event and identifying the whole range of potential predators. Game cameras allow identifying nest predators, in contrast to tracks or faeces near the nest (Benson *et al.* 2010).

Game cameras were sensitive to any movement around the nest and recorded 1 minute videos after a movement was detected during day (coloured videos) or night (by infrared illumination, black and white videos). After the first minute was recorded the camera started recording a new video until the predator had gone away or the memory card was full. Cameras were placed at a distance of ca. 50 cm from the nest and ±50 cm height above the ground by attaching them to a stick with a small tag providing information about the study. We reviewed each camera at 4 to 5-days intervals to change batteries and memory cards until a nest failed or was successful (14 days). When a nest failed the camera was moved to another nest. In total, we used 132 cameras, recording 4137 videos, each of 1-minute duration.

Data analyses

The sample unit for this study was the nest, and the response variable was nest predation (yes or no) at the 14th day after nest placement. A nest was considered predated when at least one of the eggs showed any break in the shell or had disappeared or was clearly displaced from the nest (Mezquida & Marone 2003, Batary *et al.* 2004). Five nests not found during the visits for unknown reasons were excluded from the analyses. All percentage values (%of field types and vegetation cover) were arcsine square root transformed.

To study the most important variables influencing nest predation we built averaged mixed models. We first selected candidate continuous variables by using principal component analysis (PCA) with the Varimax Normalized factor rotation. The minimum eigenvalue was 1. We selected the variable with the higher correlation value from each axis (Table 1) to reduce multicollinearity (Beja *et al.* 2013, Dormann *et al.* 2013, Barrientos & Arroyo 2014, Ponce *et al.* 2014).

Table 1. Results of the Principal Component Analysis (PCA) performed on the whole set of continuous variables. The most representative variable from each axis (marked in bold) was selected to be included in the candidate models for nest predation.

Variable	PC 1	PC 2	PC 3	PC 4	PC 5	PC 6	PC 7	PC 8	PC 9
Vegetation cover	0.73	0.08	0.02	-0.50	-0.02	-0.07	0.00	0.05	-0.18
Distance to the nearest edge	-0.01	-0.01	0.01	-0.05	**-0.97**	-0.01	-0.01	-0.04	0.01
Distance to the nearest path	0.16	0.14	0.06	0.02	-0.05	-0.02	-0.02	-0.11	0.05
Distance to the nearest river	0.03	-0.02	**0.95**	0.02	-0.03	0.02	-0.01	-0.02	-0.14
Length of edges	0.00	0.00	-0.09	0.19	0.09	0.08	-0.01	0.02	-0.05
Landscape diversity	0.08	0.32	-0.13	-0.23	0.39	-0.06	0.07	-0.24	0.17
Maximum vegetation height	0.89	0.04	0.04	-0.17	0.00	-0.05	0.19	-0.02	0.01
Mean vegetation height	**0.94**	0.02	0.06	0.14	0.02	0.06	0.05	-0.10	-0.16
Median vegetation height	0.87	0.03	0.05	0.25	0.03	0.11	-0.12	-0.12	-0.22
Length of watercourses	-0.08	0.03	-0.93	-0.01	-0.01	0.01	-0.01	0.05	-0.13
Surface of cereal crop	0.25	-0.43	0.02	0.21	0.06	0.10	-0.04	0.29	-0.73
Surface of fallows	-0.03	-0.15	-0.03	**-0.93**	-0.03	0.07	0.03	0.15	-0.03
Surface of organic vetch	0.11	**0.96**	-0.04	0.15	0.03	0.01	-0.05	0.05	0.05
Surface of organic cereal crop	0.15	-0.03	0.08	0.16	-0.02	0.06	-0.04	**-0.96**	0.01
Surface of ploughed	-0.43	-0.14	0.00	0.29	0.05	0.05	0.03	0.19	**0.80**
Surface of shrubs	-0.04	0.00	-0.01	0.05	0.00	**-0.99**	0.00	0.05	0.02
Surface of urbanized areas	0.09	-0.03	0.00	-0.02	0.02	0.00	**0.99**	0.03	0.03

We built the "*beyond optimal model*" (Zuur *et al.* 2009) with variables from the PCA plus categorical variables (location within the field -edge, interior- and field type) and different random factor structure. The error structure was binomial for the response variable. As plausible random factors we considered the site, trial and trial nested within group. We used the results from the ANOVA test in R (R Development Core Team 2013) to select the best random structure. The random structure selected was the trial nested within site. We built all possible models and selected those with an increase of<5 in the Akaike´s Information Criterion (ΔAIC) over the best model as candidate models (Burnham & Anderson 2002). Finally, we performed an average model estimation with the package MumIN (Barton 2013) in R. The final averaged model included those variables with a significant effects on the response variable: those whose confidence limits excluded zero, since they have no equivocal meaning (Delgado *et al.* 2013, Ponce *et al.* 2014, Beja *et al.* 2013).

Predator behaviour was tested by means of Chi square test. We tested which group of predators was more frequently recorded. We also tested if there were different predation patterns in relation to the field type and the location of the camera in the field (edge or interior).

RESULTS

We found 32 nests ploughed or run over by tractors. Most of them were placed in ploughed fields but there were also some nests destroyed by tractors in organic vetch crops, long-term fallows, and organic cereal crops.

We found 320 nests predated (66.3%) at the end of the study. Model averaging showed that predation was influenced by the surface of organic cereal crop and the surface of ploughed fields around the nest, the type of field where nests were placed, the location of the nest in the field, and the mean height of the surrounding vegetation (Table 2).

Table 2. Model-averaged estimates for nest predation in a dry cereal farmland area (Special Protected Area for Birds no. 139) in central Spain. The statistics given are: sum of Akaike weights of the models in which the predictor was retained (Σ), parameter estimate of the regression equation (b), standard deviation of the regression parameter (SE), lower and upper confident limits of b (Lower, Upper CI), and standardized coefficients of predictors (β). Significant predictors are marked in bold.

Scale	Predictor	Σ	b	SE	Z value	P	Lower CI	Upper CI	β
	Intercept		2.23	0.68	3.30	0.001	0.88	3.59	0.15
Landscape	**Surface of organic cereal crop**	1.00	2.26	0.72	3.17	0.002	0.83	3.69	0.16
	Surface of ploughed	1.00	2.21	0.50	4.42	0.000	1.21	3.21	0.11
	Surface of schrubs	0.47	1.52	1.28	1.19	0.234	-1.04	4.07	0.19
	Surface of fallows	0.23	0.42	0.69	0.60	0.549	-0.97	1.80	0.03
	Surface of organic vetch	0.29	-0.59	0.64	0.92	0.356	-1.87	0.69	-0.04
	Surface of urbanized areas	0.26	-0.26	2.47	0.11	0.915	-5.21	4.68	-0.06
	Distance to the nearest river	0.42	0.00	0.00	1.34	0.180	-1.5E-03	2.9E-04	-2.5E-08
	Distance to the nearest edge	0.29	0.00	0.01	0.36	0.721	-0.02	0.02	-3.0E-06
Field	**Ploughed**	0.97	-2.42	0.70	3.46	0.001	-3.82	-1.02	-0.16
	Organic cereal crop	0.97	-1.41	0.68	2.09	0.037	-2.76	-0.06	-0.09
	Cereal crop	0.97	-0.92	0.44	2.09	0.037	-1.80	-0.04	-0.04
	Location in the field (interior)	0.81	-0.55	0.27	2.07	0.039	-1.09	-0.02	-0.01
	Organic vetch	0.97	-0.63	0.54	1.18	0.239	-1.70	0.44	-0.03
Nest	**Mean vegetation height**	1.00	-0.03	0.01	4.33	0.000	-0.05	-0.02	-2.5E-05

The model predicts that the risk of predation increased in nests surrounded by more surface of organic cereal or ploughed fields. Also, predation was higher in fields with lower vegetation and in nests located near the edge of the field (70.3%, compared to a 62.3% in nests placed inside fields). The highest predation rate observed was for ploughed fields (83.3%), followed by long-term fallows (76.5%), organic cereal crops (69.9%) and vetch crops (61.4%), whereas regular cereal crops showed the lowest predation rate (43.4%, Figure 2).
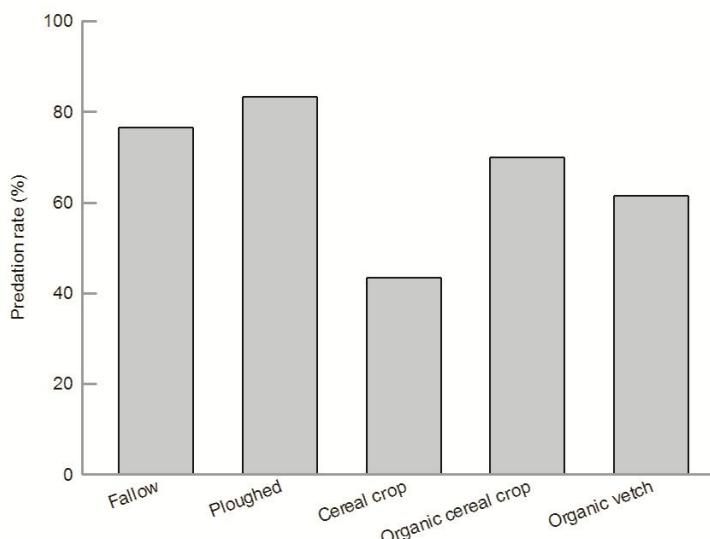
Figure 2. Mean predation rate in the different field types.

We recorded a total of 42 predation events belonging to seven wild animal species, and feral dogs. We also recorded some potential predators as feral cats, Montpellier snake (*Malpolon monspessulanus*) and rats (*Rattus spp.*) crossing in front of the camera, but none of these predated any nest. Birds were involved in more predation events than mammals (respectively, 81% and 19%, Chi-Square = 16.1, p < 0.01). The main group of bird predators were raptors. We recorded three raptor species, the Western Marsh Harrier (*Circus aeruginosus*), Montagu's Harrier (*C. pygargus*), and Common Buzzard (*Buteo buteo*), which altogether preyed upon 28 nests. Six nests were predated by Magpies (*Pica pica*). The most common mammal detected was the Stone Marten (*Martes foina*), which predated on three nests, followed by feral dogs and Edgehogs (*Erinaceus europaeus*) with two

predation events each, and Western Mediterranean Mouse (*Mus spretus*), which preyed upon one nest.

Birds did not differ on the type of field where they predated (Chi-Square = 4.8, p = 0.31), but mammals did it more intensively on fallows (Chi-Square = 11.7, p = 0.02). Also, mammals predated more intensively on nests placed on the edge than in the interior of the field (Chi-Square = 5.1, p = 0.02). There was no differential predation patterns for bird species (Chi-Square = 0.46, p = 0.50) or for raptors alone (Chi-Square = 0.18, P = 0.67).

DISCUSSION

Thirty two (6.2%) out of a total of 520 nests used in this experiment were destroyed or run over by tractors during their labours. This seems to be a common problem in farmland areas worldwide (Newton 2004, Tews *et al*. 2013). Sánchez-Oliver *et al*. (2014) and Reino *et al*. (2010) reported that respectively, 12% and 4.5% of their nests were also damaged by ploughing activities. Although most birds do not select ploughed fields for nesting, some species such as larks, curlews and lapwings prefer them over other substrates (Berg *et* al. 1992, Wilson *et al*. 1997b, Galbraith 1988, Green *et al*. 2000). Farmers plough their fields several times along the year, and farming activities are especially common in intensive agriculture areas during spring. In our study area, the common cycle of ploughing operations usually starts after harvesting in early summer, when around 20%-30% of the fields are ploughed. A second ploughing period occurs during the winter (80%-100% of the fields) and another period follows during the next spring or summer. These fields are sown in the following autumn or winter after a new ploughing operation. Although the objectives of such practices are to avoid nutrients and water loss due to the growth of weeds, the main consequence for farmland species is a marked decrease in suitable nesting habitat (Berg *et al*. 1992). However, the most common weeds found in our area during the normal nesting period (*Salsola kali*, *Heliotropium europaeum*, *Solanum nigrum*, *Datura stramonium*, etc.) develop their biological cycle in late spring and summer (Villarías *et al*. 2006), and thus do not really compete with cereal plants during

their growing or seed maturation periods. It is well known that ploughed fields with sprouted weeds are important for farmland birds during the breeding and post-fledging periods (Ponce *et al.* 2014). For that reason, we consider that current weed control during winter and early spring is enough to prevent cereal production being affected by any weed pest. We suggest that ploughing labours and, in general, field operations should be significantly reduced during this period to avoid the destruction of nests (Wilson *et al.* 2005).

The mean nest predation rate recorded in this study was 66.3%. This is an intermediate figure compared to other studies using artificial nests in the Iberian Peninsula. Sánchez-Oliver *et al.* (2014) found rates of 88.4% in open farmland habitat, though using more days of exposure, whereas Reino *et al.* (2010) obtained 49% predation rate after 15 days of exposure. Pescador & Peris (2001) found 61% predated nests after 15 days of exposure in the field.

Our results showed that different parameters related to landscape characteristics, nest location, and micro-habitat structure around the nest influenced nest predation patterns. Landscape composition played an important role, as suggested in previous studies (Reino *et al.* 2010, but see Beja *et al.* 2013). Most studies relating nest predation and landscape features compared forest plantations with open areas (e.g., Batáry & Báldi 2004), but here we have found differences within a relatively uniform landscape, the dry cereal farmland. The influential variables were the surface of organic cereal crop and ploughed fields around the nest, which showed the highest beta values in the model. Both variables had the same effect on nest predation. Predated nests had higher proportions of these two field types in a surface of 100 meters radius around the nest. However, both field types suggest contradictory explanations, since the amount of vegetation differs much between them: organic cereal fields have abundant vegetation whereas ploughed fields have very scarce or no vegetation at all. The most plausible explanation is that higher predation rates in these fields increase the likelihood of predation in surrounded fields (Wilson *et al.* 2001b). Also, predators include high quality fields (as organic cereal crops) in their home ranges, and fields joined to organic crops can attract predators (Reino *et al.* 2010).

Predation patterns also varied at the field scale, differing among specific field types in which the nest was located. The highest predation rate was found in ploughed fields, where the absence of sufficient vegetation makes nests more vulnerable. It is necessary to highlight that some ground nesting species like curlews and skylarks typically select fields with scarce vegetation for nesting (Green *et al.* 2000, Whittingham *et al.* 2002), probably because this enables them to detect approaching predators and adopt anti-predator behaviours such as distraction displays (Evans 2004). Surprisingly, nests located in long-term fallows and organic cereal crops were also highly preyed upon. Many studies have pointed out the importance of uncultivated and organic crop fields for biodiversity and for bird nesting (O'Connor & Shrubb 1986, Wilson & Browne 1993, Brickle *et al.* 2000, Hole *et al.* 2005). In our study, both field types were favoured by the AES, and thus agri-environmental measures did not succeed in protecting nests from high predation rates, in spite of theoretically providing more nest concealment opportunities. As explained above for the landscape structure in circles of 100 m radius around nesting sites, fallows and organic cereal fields probably attracted predators due to the higher prey abundance and diversity contained in them, and so the probability of nests being preyed upon increased in these substrates. Vegetation height was a significant factor in the model describing predation probability, but the vegetation grown in fallows and organic fields did not reach a height sufficient to offer optimal nest concealment opportunities. The same result was reported by Pescador & Peris (2001), who also suggested that fallows attracted predators. The same result was reported by Pescador & Peris (2001) in Spain. They suggest that fallows are scarce in their study area and predators are attracted to them. In the place where we carried out the experiment both fields were present. However, they are scarce at higher scale.

Our results support the idea of "*ecological trap*" suggested by other researchers to explain high predation rates in high quality fields (Chamberlain *et al.* 1995, Donald 1999, Evans 2004, Sokos *et al.* 2013). It is well known that long-term fallows and organic cereal crops are highly selected by birds during the nesting period (Beja *et al.* 2013, Newton 1998, Wilson *et al.* 2001a, Hole *et al.* 2005). A balanced pay-off might exist between the high amount and quality of food available in these fields and the high predation risk on them. Nests in fallows can

suffer relatively high predation rates, but on the positive side, the abundant food available on them allows chicks to grow and survive better compared to other fields (Hole *et al.* 2005).

The lowest predation rates were detected in regular cereal crops. These fields had the tallest vegetation, and thus offered the best concealment opportunities for nesting. In our study area, cereal crops were selected by Great bustards as preferred substrates for nesting (Magaña *et al.* 2010). Other authors have also found that nest concealment may decrease the probability of nest predation (Rangen *et al.* 1999, Beja *et al.* 2013). Besides, in a natural situation the density of nests is probably lower in cereal fields than in other substrates (Wilson *et al.*1997b), due to the low food availability, which plays an important role for nest site selection (Kragten & De Snoo 2007, Kragten *et al.* 2008).

Most of previous nest predation studies were carried out comparing edges with forested areas (e.g., Benson *et al.* 2010). In our study, nests placed near the field edge were more intensively preyed upon than those located inside the fields. There are different plausible reasons for this result. First, field edges are frequently used by birds during the breeding season for nesting, which makes them attractive and profitable sites to predators searching for food (Gates & Gysel 1978, Chalfoun *et al.* 2002). Second, it is known that predators use linear structures to move between areas (Bider 1968), although this behaviour is more common in mammals tan in birds.

Finally, camera trapping results showed that different predators selected different habitats to search for food (Bayne *et al.* 1997, Benson *et al.* 2010). In our study area cameras detected four times more birds than mammals, and the most frequent bird predators were raptors. Birds, and particularly raptors are well known nest predators (Opermanis 2001, Batary *et al.* 2004, King & DeGraaf 2006, Purger *et al.* 2008). Benson *et al.* (2010) found that raptors concentrated their searching effort near field margins. In our study, neither all birds nor the group of raptors showed any preference for field margins. In contrast, mammals showed a higher predation effort near edges than inside fields. Food searching patterns differ widely between mammals, which use olfactory cues and typically follow the same routes like paths, field borders, or their own previous tracks, and birds,

which search for nests by means of visual stimuli and cross fields flying without obvious obstacles (Rangen *et al.* 1999).

MANAGEMENT IMPLICATIONS

Nest destruction by tractors that plough fields during the breeding period is a problem in our study area. Since other studies have also identified this problem, we believe it is important that future AES consider including some restrictions on agricultural machinery to prevent direct nest destruction. Measures should also allow some vegetation growth to favour birds nesting in ploughed fields (Donald *et al.* 2002).

Predation is considered one of the causes of the recent declines observed in ground-nesting populations of farmland species. The origin of this high predation is related to agricultural intensification (Newton 2004). The impact of predators on farmland species can be counteracted in different ways. One possibility is predator control (Suárez *et al.* 1993), which has been proved effective for several species (Thirgood *et al.* 2000, Fletcher 2006). However, this possibility does not solve the problem in the long term. Also, in our study the main predators were raptors, which are strictly protected by national and international law. A second possibility is through habitat management tools (Evans 2004). In this case, the correct design of AES is essential for reducing nest predation of ground-nesting species.

The results from our study provide strong evidences that predation rates are influenced by factors acting at landscape, field and nest-site scale. At the landscape scale, we found predation increases for two situations strongly differing in vegetation cover around the nest site: areas where the surface of organic cereal were dominant, and those where ploughed land was dominant. This was corroborated at the field scale, where nest predation was higher in fallows and organic cereal crops, two of the field types promoted by AES. One possible way to minimize this predation increase would be to disperse these fields promoted by AES in order to avoid the development of isles of high quality, where predators are expected to focus their hunting efforts. In this way, the food-related benefits of

agri-environmental fields would be the same, but would not be counteracted by high predation rates. In addition, to prevent a higher predation rate on nests near field borders, we suggest that fields from AES should be large, allowing birds to find appropriate nesting sites far from edges (Bayne *et al*. 1997).

Finally, studying the composition of the predator community is necessary to understand the mechanisms underlying predation rates in different scenarios (Benson *et al*. 2010). In areas where birds are the main nest predators, AES measures should focus on increasing vegetation height, to maximize the offer of nesting sites with appropriate concealment against aerial predators (Davis 2005).

All these measures aiming at reducing nest predation should be taken into account together with measures enhancing food abundance when designing AES programs in dry cereal farmland areas.

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DISCUSIÓN GENERAL

En esta tesis doctoral se han estudiado dos aspectos fundamentales para la conservación de los ambientes agrícolas, como son la eficacia de la señalización en tendidos eléctricos para reducir la mortalidad directa de las aves debida a las colisiones y los efectos del manejo del hábitat para revertir el proceso de intensificación agrícola. El objetivo último de ambas actuaciones es el mismo: mejorar la sostenibilidad de las zonas agrícolas y de la biodiversidad que mantienen, para que la conservación de las estepas agrícolas y sus comunidades de plantas y animales sean viables a largo plazo. Para conseguir ese objetivo, en esta tesis se han estudiado la intensificación agrícola y los tendidos eléctricos, que junto a otras infraestructuras asociadas al desarrollo humano (carreteras, edificios, etc.), la presión cinegética y otras causas locales, son responsables del declive de las aves esteparias (Morales *et al* 2005).

Debido a la desaparición acelerada de la fauna esteparia, los planes de recuperación y conservación de fauna y flora son aplicados en zonas remotas con poco desarrollo (por ejemplo: Lagunas de Villafáfila, Zamora) en los que existe fauna y flora de interés para la conservación, pero donde los impactos antropogénicos son escasos. Para estudiar el impacto de la intensificación agrícola y del desarrollo de los tendidos eléctricos ha sido preciso seleccionar una zona en las afueras de una ciudad como Madrid (con más de3 millones de personas), en la que la intensificación agrícola se pueda revertir o atenuar mediante la aplicación de un Plan de Medidas Agroambientales. La investigación presentada en esta tesis es, por tanto, muy infrecuente, tanto por la escala de los experimentos (señalización de tendidos, plan de medidas agroambientales, años de trabajo) como por desarrollarse en una zona peri-urbana, en la que la densidad de tendidos eléctricos es propia de una gran urbe, aunque a pesar de ello, sigue manteniendo una rica comunidad de aves esteparias (incluyendo una importante población de Avutarda común *Otis tarda*, Martín 2008). Las zonas degradadas medioambientalmente en los alrededores de las grandes ciudades son frecuentes, pero investigar en ellas la eficacia de las medidas agroambientales o el impacto de los tendidos eléctricos suele ser difícil, debido a que no tienen la suficiente riqueza de especies y/o sus poblaciones son inviables por su pequeño tamaño. Los

tendidos eléctricos son comunes en estos ambientes de la zona centro peninsular, tanto en la Comunidad de Madrid como en la provincia de Guadalajara. No existe ningún lugar habitado por la avifauna estepria en estas zonas en la que falte un tendido eléctrico, aunque la densidad de líneas eléctricas, lógicamente, varía entre zonas (Martín 2008). En esta tesis, la zona de estudio y las especies que la habitan se encuentran entre las que mayor cantidad de individuos contienen de todo el rango de distribución en España y otros países de Europa (en el caso de la avutarda). Aunque el resultado general de las medidas adoptadas puede ser positivo tras haberse estudiado su efectividad en esta tesis, ambas cuestiones continúan sin tener una solución plenamente satisfactoria.

Evaluación de la mortalidad de los tendidos y la eficacia de las medidas compensatorias para reducirla

La interacción entre las aves y los tendidos eléctricos y la forma de mitigar la mortalidad asociada a la colisión han sido muy estudiadas durante los últimos 50 años (ver revisión en Barrientos *et al*. 2011), a pesar de lo cual no se han obtenido resultados plenamente satisfactorios en ningún trabajo, en cuanto a la eliminación total de este factor de mortalidad. Este hecho pone de manifiesto la complejidad del problema que suponen estas infraestructuras para las aves. A esto se debe añadir la falta de estandarización de los métodos de recogida de la información y la falta de robustez en algunos de los análisis efectuados (Bevanger 1999). Además, existen varios sesgos que pocas veces se han considerado en este tipo de estudios, por lo que sus conclusiones y extrapolaciones a otras zonas se deben tomar con cierta cautela.

El trabajo mostrado en esta tesis permitió desarrollar una metodología para calcular de qué manera se subestima la cantidad real de aves muertas en tendidos eléctricos y los factores implicados (capítulo 1), así como estimar la mortalidad producida en las líneas eléctricas y estudiar la eficacia de la colocación de dispositivos anticolisión (capítulo 2).

Los estudios sobre mortalidad en tendidos eléctricos tienen como primer objetivo conocer la magnitud del problema que generan la colisión y/o la electrocución en una zona determinada. Por otro lado, cuando se señaliza un

tendido eléctrico el objetivo es reducir la mortalidad de aves. Según se ha puesto de manifiesto en esta tesis, es fundamental llevar a cabo varios tipos de experimentos (desaparición de cadáveres y detectabilidad según el observador), puesto que existen sesgos que hacen variar en gran medida los resultados obtenidos.

En nuestro caso, encontramos diferencias en la desaparición de cadáveres según el tamaño de los ejemplares. Aunque se ha propuesto una ecuación general válida para los diferentes tamaños (Figura 4, capítulo 2), es probable que existan grandes variaciones en la tasa de desaparición respecto a otros hábitats. Aunque el objetivo principal de ese capítulo no incluía este aspecto, sí debe considerarse a la hora de extrapolar las ecuaciones obtenidas a otros lugares.

En estudios científicos y técnicos sobre interacciones entre aves e infraestructuras suelen participar personas con muy diversa formación en la búsqueda de cadáveres. Tal y como muestran los resultados de esta tesis, sería recomendable cambiar este aspecto, ya que los resultados obtenidos podrían estar sesgados, aumentando el número de ejemplares localizados por observadores más experimentados (Figura 5, capítulo 2). En este caso, los resultados obtenidos no serían válidos si no se corrigen por la experiencia en la localización e identificación de los restos. Por tanto, existen tres opciones para evitar este sesgo. Una es que los participantes en este tipo de proyectos tengan la misma formación de partida. Otra es realizar un número alto de sesiones en búsqueda de cadáveres bajo los tendidos eléctricos previos al comienzo de los muestreos para el estudio. Por último, otra posibilidad es que se lleven a cabo experimentos similares a los de esta tesis doctoral y se apliquen las correspondientes correcciones. Sin duda, este último caso aportaría un valor extra a cualquier trabajo de este tipo.

La investigación llevada a cabo para conocer la eficacia de la señalización en tendidos eléctricos (capítulo 2) ha permitido aplicar los valores de desaparición de cadáveres del capítulo 1. Los resultados obtenidos mediante el número de cadáveres encontrados fueron muy diferentes de los obtenidos mediante las estimas aplicando las correcciones necesarias (Tabla 4, capítulo 2).

Pudimos estimar de una forma precisa cuántas aves mueren en los tendidos eléctricos de zonas agrícolas del centro peninsular. Además, se obtuvo un listado de especies realmente amplio (Tabla 3, capítulo 2), algunas de ellas protegidas, como la avutarda, Sisón común (*Tetrax tetrax*), Buitre negro (Aegypius monachus) o Aguilucho lagunero occidental (Circus aeruginosus). La señalización fue eficaz, incluso cuando se desciende a nivel de especie. Por ejemplo, ésta fue más efectiva para el sisón y, algo menos para la avutarda. Es necesario recordar que ambas se encuentran amenazadas según diferentes catálogos, tanto a nivel regional como nacional, son especies representativas de los ambientes agrícolas del centro peninsular y parecen proclives a las colisiones con tendidos eléctricos (Alonso *et al*. 1994, Barrientos *et al*. 2011, Ferrer 2012). Por tanto, estudiar la eficacia de la señalización ha permitido calcular la magnitud real sobre estas y otras especies que viven en medios agrícolas.

Aunque la señalización ha sido eficaz a la hora de disminuir la mortalidad de las aves, no se ha eliminado completamente este factor de mortalidad. Es necesario estudiar cuál es la influencia de los tendidos eléctricos en la viabilidad poblacional de especies amenazadas (Bevanger 1999) antes (Martín 2008) y después de la señalización. En este sentido cabe decir que el equipo de investigación en el que se ha desarrollado esta tesis doctoral dispone de una serie larga de años de censos para la avutarda, además de numerosos datos sobre ejemplares marcados muertos en los tendidos eléctricos de toda España. Estas bases de datos tienen un valor creciente y permitirán en el futuro llevar a cabo esos análisis post-señalización, así como otros sobre la localización de lugares concretos donde la mortalidad es más elevada dentro de un tramo de tendido (puntos de riesgo máximo o puntos negros). De esta manera, si se produjera una nueva señalización o refuerzo de la ya existente, se podrían dirigir los esfuerzos hacia esos lugares, sin tener que emplear altos presupuestos económicos.

El 29 de Agosto de 2008 se aprobó el real Decreto 1432/2008, por el que se establecen medidas para la protección de la avifauna contra la colisión y la electrocución en líneas eléctricas de alta tensión. Dicho decreto establece la obligatoriedad de marcaje o modificación de tendidos eléctricos peligrosos en zonas protegidas. Sin embargo, la obligación se refiere exclusivamente a la

electrocución de aves, sin considerar la colisión, cuya mitigación será voluntaria. Tal y como se ha puesto de manifiesto en esta tesis y otras investigaciones (Ferrer 2012), la colisión es un grave problema para varias especies de aves esteparias. No se entiende que en el real Decreto no se haya incluido la colisión de aves como una amenaza de similar magnitud a la producida por la electrocución en líneas eléctricas de zonas protegidas (Íñigo *et al*. 2010). Sería necesario hacer una revisión del Real Decreto para subsanar este grave error.

Además, el real decreto deja fuera del ámbito de aplicación de las medidas todas las zonas que no estén incluidas en zonas de protección. En la Comunidad de Madrid existen varios lugares que quedan fuera de esas zonas protegidas, pero que albergan importantes grupos de avutardas y sisones (más susceptibles que otras especies a la colisión), como son Campo Real, Fuentidueña-Estremera de Tajo, o Torrejón de Velasco.

Evaluación de la eficacia de las medidas agroambientales para reducir los efectos de la intensificación agrícola

Las zonas agroesteparias suponen una gran parte de la superficie europea y española, por lo que su gestión afecta a la conservación de multitud de organismos, tanto plantas como animales. Por otro lado, los cambios producidos en la agricultura durante las últimas décadas han desembocado en severos problemas ambientales para distintos grupos de seres vivos, hasta el punto en que se considera a la intensificación la principal causante de dichos problemas. Para contrarrestar ese efecto negativo se ha actuado mediante Programas de Medidas Agroambientales a nivel continental y estatal (Carricondo *et al*. 2012), aunque los resultados varían enormemente entre regiones (Kleijn & Sutherland 2003).

Los resultados proporcionados en esta tesis apoyan la idea general de que el manejo del hábitat a través de medidas agroambientales puede llegar a aportar beneficios ambientales a los sistemas agrícolas (Kleijn *et al*. 2006). Se han implementado medidas para favorecer el tamaño y composición de las especies de plantas, artrópodos (capítulo 3) y aves (capítulos 4 y 5). Sin embargo, la eficacia de cada una de las medidas adoptadas no es igual en cada grupo de organismos. En la presente tesis se pone de manifiesto la necesidad de reducir la intensificación

agrícola para favorecer a los organismos de los sistemas cerealistas de secano mediterráneos (Sans *et al.* 2013). La cuestión más complicada es, seguramente, decidir de qué manera se debe llevar a cabo ese proceso para revertir la situación actual sin perjudicar la producción agrícola, tendente al incremento (Wilson *et al.* 2009, Pretty *et al.* 2010). Existen numerosas medidas que, además, varían en la forma de implantación en cada uno de los países (Kleijn *et al.* 2006) e incluso a escala regional dentro de cada uno. Basta ver las medidas desarrolladas por la Comunidades Autónomas (Carricondo *et al.* 2012).

En el capítulo 3 de la tesis se lleva a cabo un análisis de la eficacia del manejo en parcelas de cultivo de cereal. Concretamente se estudia la respuesta de plantas silvestres y artrópodos al cultivo ecológico de cereal en comparación con el manejo convencional. El manejo ecológico del cereal fue positivo para ambos grupos (Tablas 3 y 5), a pesar de que el manejo ecológico sólo se ha producido durante un año, cuya consecuencia es que los herbicidas empleados en años anteriores persisten. Se incrementaron todos los valores analizados respecto a las plantas silvestres (Tabla 3) y al comparar los resultados con los de otras regiones se obtiene que el incremento en parámetros de vegetación es mucho mayor en nuestra zona de estudio. La Región Mediterránea es más rica en plantas silvestres, la agricultura de nuestra zona no es tan intensiva como en otros lugares y las parcelas son de pequeño tamaño. Todo ello hace que se den las condiciones apropiadas para una mayor proliferación de plantas silvestres. También aumentaron la mayor parte de los parámetros relacionados con los artrópodos (Tabla 4). Tanto la abundancia, como la riqueza y la biomasa (esta ligeramente). Lógicamente, esto se relaciona con la eliminación del uso de biocidas en las parcelas cultivadas de forma ecológica. Sin embargo, una consecuencia del manejo ecológico fue la proliferación de varios grupos de artrópodos, lo que hizo que la diversidad en siembras convencionales fuera mayor que en la siembras cultivadas de forma ecológica.

El aumento de esos parámetros mediante el cultivo ecológico de cereal ha producido consecuencias adversas en la depredación de nidos artificiales (Figura 2, capítulo 5). Los nidos localizados en parcelas cultivadas de forma ecológica y aquellos que estaban rodeados por mayor superficie de cultivo de cereal ecológico

vieron incrementada su tasa de depredación (Tabla 2). En cambio, las siembras de cereal convencional no produjeron efectos positivos en la mayoría de los parámetros estudiados sobre plantas, artrópodos y aves (capítulo 3, capítulo 4), pero sufrieron las menores tasas de depredación de nidos (Figura 2, capítulo 5). Así pues, se genera un conflicto entre los beneficios ambientales y la depredación de nidos. Tal y como han mostrado otros autores (Redisma *et al.* 2006, Armengot *et al.* 2011), existe un gradiente en el manejo del cultivo de cereal, tanto convencional como ecológico. Para compaginar ambas cuestiones, y obtener resultados positivos en ambos casos, sería necesaria la adopción de una solución de compromiso. Es decir, reducir el manejo intensivo en las siembras de cereal convencional y no llegar al extremo del cultivo ecológico. De esta forma sería esperable un aumento del valor ecológico de las siembras convencionales, aunque también un incremento en la tasa de depredación, lo contrario para las siembras manejadas de una forma más extensiva. Esta hipótesis no ha sido comprobada en esta tesis, pero los resultados obtenidos permiten su formulación para comprobar su veracidad en futuros experimentos.

El análisis de los parámetros durante todo el ciclo anual de las aves reflejó que el manejo de parcelas incluidas en el programa de medidas agroambientales resultó positivo para todas las épocas del año (Tabla 2, capítulo 4). Sin embargo, otras variables no incluidas en el programa también resultaron importantes, probablemente debido a que la zona de estudio no está especialmente intensificada (Concepción *et al.* 2008, Concepción *et al.* 2012). Por tanto, uno de los primeros resultados destacables de este capítulo es la necesidad de considerar variables a una escala mayor que las propias medidas agroambientales implantadas.

Los dos grupos de modelos retuvieron básicamente similares predictores con independencia de la dificultad de medir las variables que los integraban (modelos complejos vs modelos sencillos). Sin embargo, los modelos complejos que incluían variables costosas de medir (Tabla 5) tuvieron en general mejores resultados a la hora de predecir la respuesta de las aves ante el manejo agrícola (Tabla 4) que los modelos sencillos. Los modelos complejos permitieron entender de una manera más precisa los factores que subyacen a esas respuestas. Hay varios

casos en los que estos modelos revelaron la importancia de las variables de comida, más que la estructura o la superficie de la parcela, que no habrían salido a la luz si se hubiera empleado el conjunto de modelos de hábitat (modelos sencillos). Sin embargo, el esfuerzo y dinero necesarios para llevar a cabo los modelos complejos son altos en comparación con los modelos sencillos de hábitat. Sugerimos que el desarrollo de los programas de medidas agroambientales que se lleven a cabo en el futuro consideren la posibilidad de incluir un presupuesto específico para poder registrar en campo las variables necesarias para el cálculo de los modelos complejos, de manera que sea posible una evaluación científica más precisa de la eficacia de dichos programas.

El trabajo de campo desarrollado ha puesto de manifiesto la importancia de semillas y artrópodos para la dieta de las especies de aves esteparias durante el invierno. Aunque este hecho ya ha sido propuesto en otros estudios (Evans *et al.* 2011), es necesario destacar que las semillas empleadas en el programa de medidas agroambientales de esta tesis doctoral no estuvieron sometidas a ningún tratamiento fitosanitario, gracias a lo cual no se observaron los daños a las aves descritos en otros estudios (López-Antia *et al.* 2015).

La medida más efectiva para las variables medidas en el capítulo 4 fue el barbecho de larga duración (o sea, la retirada de la producción de una parcela agrícola) debido a la gran cantidad de alimento que aporta mediante semillas, artrópodos y plantas (Chamberlain *et al.*1999, Henderson *et al.* 2000, Lapiedra *et al.* 2011). Sin embargo, también fue uno de los tipos de parcela más depredados, casi tanto como las parcelas labradas (Figura 2, capítulo 5). El hecho de que parcelas de alta calidad (capítulo 3 y 4) sean notablemente más depredadas que las siembras convencionales sugiere la posibilidad de la presencia de trampas ecológicas (Donald 1999, Evans 2004), debido a un fenómeno de atracción de depredadores a los lugares con mayor densidad de comida (aves, micromamíferos, nidos, etc.). Los planes de conservación de aves ligadas a estos medios deben considerar la interacción de ambos efectos a la hora de implementar los barbechos de larga duración. Según los resultados obtenidos, las parcelas de barbecho y de cultivo ecológico de cereal deberían dispersarse en grandes áreas. De esta forma se mantendrían los efectos positivos de las medidas y se reduciría, probablemente, la

depredación (Bayne *et al.* 1997). Existe literatura sobre el impacto que tiene la concentración de parcelas de alto valor ecológico en una matriz agrícola (por ejemplo, bosques isla en las comunidades de aves, Santos *et al* 2002), pero en el caso de las aves esteparias se desconoce cómo influye la dispersión de parcelas valiosas en su comportamiento y la evolución de sus poblaciones.

El cultivo de leguminosa también fue importante, tanto la biomasa de semillas de veza y los artrópodos como la superficie cultivada presentes en todas las épocas del año y para muchas de las variables respuesta consideradas (Tabla 3, capítulo 4). Es de sobra conocido que las leguminosas son una fuente importante de alimento para las aves esteparias (Bretagnolle *et al.* 2011, Bravo *et al.* 2012). Lo que no se había estudiado hasta la fecha es que la tasa de depredación producida en este cultivo es la más baja respecto a cualquier otra medida agroambiental aplicada (Figura 2, capítulo 5). Además, el beneficio de la semilla no tratada hace de esta medidas muy útil y necesaria para las aves de zonas agrícolas.

Las parcelas labradas aparecieron también como factor importante en los análisis de abundancia y riqueza de aves (Tabla 2, capítulo 4). Especies como la alondra común o la avefría europea prefieren este tipo de sustrato para alimentarse debido a que su estrategia frente a los depredadores implica tener un gran campo visual y poder huir antes de que el depredador esté cerca (Butler *et al.* 2005). Sin embargo, este tipo de parcela sufrió la mayor tasa de desaparición, relacionada con la escasez de vegetación donde poder cobijar los nidos (Figura 3, capítulo 5). Determinadas aves, como las mencionadas anteriormente, emplean este tipo de parcelas también durante la época de nidificación, basando su estrategia antidepredadora en el mismo procedimiento descrito anteriormente (Whittingham *et al.* 2002). A la alta tasa de depredación hay que sumarle el riesgo de ser destruidos durante las labores agrícolas de los tractores. Una especie que nidifique en este sustrato tiene altas probabilidades de que su huevos no puedan llegar a eclosionar. Es necesario abordar este problema con rapidez debido a su alto impacto sobre las aves ligadas a medios abiertos y las consecuencias directas hacia las poblaciones. En esta tesis se proponen varias medidas que podrían resultar eficaces. Sería necesario restringir las labores agrícolas durante la época de nidificación, lo que evitaría la destrucción directa de nidos. Además, sería

recomendable favorecer un cierto grado de desarrollo vegetal el cual ayudaría a que las aves nidificantes en este sustrato dispongan de lugares más protegidos y mayor alimento. Por último, en algunas zonas existe un ciclo agrícola demasiado acoplado. Un año la mayor parte de las parcelas están cultivadas, y al siguiente todas ellas son parcelas labradas. Si se desacoplara el ciclo agrícola es posible que ello redundase en menores tasas de depredación y en un aumento importante de la cantidad de alimento disponible para las aves, no sólo a escala de parcela, sino también de paisaje.

La colocación de los nidos en el borde y el interior de la parcela evidenció una depredación diferencial (Tabla 2, capítulo 5). Los nidos del borde de la parcela tuvieron mayor riesgo de ser depredados. Este hecho, junto con los resultados de la estructura de la vegetación (menor depredación cuanto más alta es la vegetación), y la determinación de los grupos de depredadores, ayudaron a explicar los patrones obtenidos. En nuestro estudio, la mayor parte de los eventos de depredación los protagonizaron las aves. Éstas no depredaron más sobre un tipo de parcela concreta, ni se detectaron diferencias respecto a la localización del nido dentro de la parcela. El motivo es que la estrategia de búsqueda de alimento de las aves (sobre todo las rapaces registradas) se basa en prospectar el territorio en vuelo y detectar a sus presas de forma visual. Los mamíferos, en cambio, sí depredaron más sobre los barbechos y en nidos localizados en el borde de la parcela. Esto se debe a que los mamíferos se basan en el olfato para detectar a sus presas y, además, utilizan las estructuras lineales para moverse entre zonas. El control de depredadores no es recomendable en nuestra zona de estudio. A pesar de no tener información sobre este parámetro, la ausencia de zorros (Vulpes vulpes) nos indica que ya existe un control de depredadores en la zona. Por otro lado, los aguiluchos fueron los depredadores que más aparecieron en las cámaras. Ambas especies de aguiluchos (cenizo y lagunero) están estrictamente protegidas. El aguilucho cenizo está amenazado y se están llevando a cabo medidas para favorecerlo, como el salvamento de nidos o el retraso en la recogida de cereal. Sugerimos que se haga un manejo del hábitat para dificultar el acceso de los depredadores a los nidos mediante el desarrollo de la vegetación y, por tanto, la mayor ocultación de los nidos (Davis 2005).

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CONCLUSIONES

1 Los estudios sobre mortalidad de aves en tendidos eléctricos llevan asociados varios sesgos que infravaloran la cantidad real de aves muertas. Se desarrollan y proponen varios índices de corrección para la desaparición de cadáveres y para su detectabilidad en función del tamaño de las aves y de la experiencia de los observadores, y se proponen las frecuencias de revisión de tendidos más adecuadas.

2 La mortalidad de aves colisionadas contra tendidos eléctricos supone un grave problema en las estepas cerealistas del centro peninsular. La señalización de tendidos eléctricos reduce la mortalidad de forma significativa, aunque no la elimina, desconociéndose además las implicaciones sobre la viabilidad poblacional en de especies amenazadas.

3 El manejo del hábitat agrícola mediantes medidas agroambientales beneficia a plantas silvestres, artrópodos y aves. Sin embargo se debe profundizar en el efecto de las medidas implementadas sobre las especies amenazadas.

4 El cultivo de cereal ecológico beneficia a plantas y artrópodos, aunque produce un incremento en la tasa de depredación de nidos. La reducción de la intensificación en siembras convencionales podría proporcionar mayor cantidad de alimento, así como otros beneficios, a los grupos considerados. Sin embargo, es probable que repercuta de manera negativa sobre la tasa de depredación de nidos.

5 Las aves esteparias se ven favorecidas durante todo el ciclo anual por las medidas agroambientales. Sin embargo, otras variables relacionadas con el paisaje agrícola también son influyentes. El desarrollo de modelos complejos, aunque costosos, permite conocer los factores subyacentes a la respuesta de las aves de una forma más precisa que modelos más sencillos. Así, las aves esteparias se ven favorecidas más por la cantidad de alimento presente que por la composición del mosaico de cultivos.

6 La depredación de nidos está influida por variables a escalas de paisaje, parcela y lugar de nidificación. Las medidas agroambientales no logran reducir la tasa de depredación de nidos, debido a que las medidas atraen a los depredadores. Para reducir la tasa de depredación se debe incrementar la altura de la vegetación y distribuir las parcelas con medidas de forma más dispersa.

7 España tiene un mayor compromiso que otros países de Europa en la conservación de los hábitats agrícolas y su biodiversidad asociada. Alberga algunas especies ausentes o escasas en otros países y el manejo del hábitat en nuestras latitudes tiene consecuencias en la conservación a escala continental.

ANEXO 1

Programa de Medidas Agroambientales del Área Importante para las Aves Talamanca-Camarma.

Programa de Medidas Agroambientales

del Área Importante para las Aves

Talamanca-Camarma

Dirección de contacto:

Carlos Ponce Cabas

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Concesionaria del Estado

INTRODUCCIÓN

Las áreas de carácter estepario ocupan una buena parte del territorio de las provincias de Madrid y Guadalajara. La mayoría de ellas tienen su origen en las prácticas agrícolas y ganaderas que durante milenios han trasformado los bosques primitivos en extensos campos dedicados a cultivos de secano o pastizales.

La agricultura extensiva tradicional permitía la coexistencia de explotaciones agrarias dedicadas al cultivo de herbáceos de secano con una rica biodiversidad, en especial, con importantes comunidades de aves esteparias. El reciente proceso de intensificación de la agricultura ha modificado el equilibrio existente entre hombre y aves en las llanuras, poniendo en peligro a gran parte de las especies de aves esteparias.

Las importantes poblaciones de diferentes especies de aves esteparias que todavía habitan en la península Ibérica pueden considerarse únicas a escala europea, por lo que su conservación depende en gran medida del mantenimiento de los ecosistemas agroesteparios ibéricos.

La pérdida y fragmentación del hábitat agroestepario ocasionada por la construcción de grandes infraestructuras es otra de las principales causas de disminución de la superficie de ecosistemas agroesteparios. Los efectos negativos de dichas intervenciones humanas se derivan, por una parte, de la pérdida neta de superficie disponible para las especies, y por otra, de la fragmentación del hábitat, que resulta dividido en unidades cada vez menores.

Con el fin de compatibilizar la actividad agrícola con la conservación de la naturaleza, y para compensar la pérdida de hábitat ocasionada por la construcción y explotación de las autopistas R-2 y M-50, se propone el presente Programa de Medidas Agroambientales del Área Importante para las Aves "Talamanca-Camarma". Este Programa está incluido en el *Proyecto de medidas preventivas, correctoras y compensatorias de la afección de la M-50 (tramo M-607 / N-IV, subtramo N-I / N-II) y de la Autopista de peaje R-2 a la población de avutardas y otras aves esteparias de la IBA "Talamanca-Camarma", y al LIC "Cuenca de los ríos Jarama y Henares".*

OBJETIVOS DEL PROGRAMA

El objetivo principal de este Programa es compatibilizar la conservación de las poblaciones de aves esteparias con la explotación agrícola de secano dentro de las provincias de Madrid y Guadalajara. A través de un sistema de primas económicas se pretende beneficiar a los agricultores que utilicen métodos de producción agraria compatibles con la conservación de la biodiversidad.

ÁMBITO DE APLICACIÓN

Las actuaciones se llevarán a cabo en un total de siete zonas (ver anexo 1) situadas dentro del Área Importante para las Aves denominada "Talamanca-Camarma", en las provincias de Madrid y Guadalajara. Seis zonas estarán situadas en la provincia de Madrid, prácticamente todas ellas dentro de la Zona de Especial Protección para las Aves "Estepas cerealistas de los ríos Jarama y Henares", y una en la provincia de Guadalajara, dentro de la ZEPA "Estepas cerealistas de la Campiña". Se han seleccionado estas siete zonas, por ser las que cuentan con mayor diversidad y tamaño de poblaciones de aves esteparias y por estar directamente afectadas por la construcción de las autopistas R-2 y M-50.

BENEFICIARIOS

Podrán acogerse voluntariamente al Programa de ayudas todos los agricultores con tierras de secano dedicadas al cultivo de herbáceos incluidas en el ámbito de aplicación del Programa. En el caso de aquellos agricultores que explotan tierras en régimen de arrendamiento o aparcería tendrán que actuar de acuerdo con el propietario.

Este Programa va dirigido a las superficies dedicadas al cultivo de herbáceos de secano. Se excluyen cultivos de regadío, cultivos leñosos y superficie de erial, pastizal o arbolado, así como terrenos improductivos. Sin embargo, es perfectamente compatible con los programas de ayudas agroambientales derivados de los Reglamentos 2078/92 y 1257/1999 de la CEE y del Real Decreto 4/2001.

Los beneficiarios podrán elegir entre las cinco medidas que aparecen descritas a continuación.

COMPROMISOS DE LAS MEDIDAS

Medida 1: Mejora y mantenimiento del barbecho tradicional

Para el conjunto de parcelas acogidas a esta medida, que estarán destinadas a barbecho, el agricultor se comprometerá a:

- Mantener los rastrojos sin alzar desde la cosecha de cereal precedente, en el mes de julio, hasta el 1 de enero. A partir de esta fecha el agricultor podrá labrar los barbechos sin aplicar productos fitosanitarios ni ninguna otra sustancia química hasta el 31 de marzo
- Nuevamente durante los meses de abril, mayo y junio no podrá realizar ninguna labor agrícola sobre los barbechos acogidos a esta medida

Medida 2: Barbecho semillado con leguminosas

Para las parcelas acogidas a esta medida, el agricultor se comprometerá a:

- Siembra de leguminosas (veza, yeros, alfalfa, guisantes, garbanzos…) sobre parcelas destinadas a barbecho
- Preparar el terreno correctamente para el buen desarrollo de las plantas de leguminosa
- Comunicar la fecha de siembra al responsable del Programa de Medidas Agroambientales al menos 5 días antes de realizarla.
- No emplear más del 20% de semilla de cereal junto con la semilla de leguminosa
- No utilizar semillas tratadas o blindadas para la sementera.
- La siembra debe ser realizada en el mes de octubre
- Enterrado de dichos barbechos semillados no antes del 10 de julio
- No utilizar abonos ni productos fitosanitarios durante el período de duración del barbecho semillado

Medida 3: Retirada de la producción de tierras durante el periodo de duración del programa (Máximo de 5 años para los acogidos en el ciclo agrícola 2006-2007)

Para las parcelas acogidas a esta medida, el agricultor se comprometerá a:

- No realizar labores agrícolas durante el periodo establecido de retirada de la producción
- No utilizar productos fitosanitarios durante el período de retirada
- No quemar el barbecho durante el período de retirada
- Para acogerse a esta medida es necesario entregar una declaración o comprobante del uso agrícola de la tierra durante los últimos 3 años.

Medida 4: Rotación de cultivos trigo- girasol

Para las parcelas acogidas a esta medida, el agricultor se comprometerá a:

- Mantener los rastrojos procedentes de la siembra del cereal precedente, sin alzar hasta al menos el 1 de enero
- La siembra de girasol será efectuada en un intervalo de tiempo que comprende desde el 1 de enero hasta el 31 de marzo, sin poderse prorrogar con posterioridad a esta ultima fecha
- Preparar el terreno correctamente para el buen desarrollo de las plantas de leguminosa
- Comunicar la fecha de siembra al responsable del Programa de Medidas Agroambientales al menos 5 días antes de realizarla
- La siembra se realizará en cantidades no inferiores a 3,250 Kg. por hectárea o 45.000-50.000 plantas por hectárea, con una separación entre líneas de cultivo de aproximadamente 70 cm
- Los agricultores acogidos a esta medida se comprometerán a no utilizar herbicidas en el cultivo del girasol
- Las labores de triturado y enterrado del cañote del girasol no podrán ser efectuadas antes del 30 de septiembre

Medida 5: Cultivo de cereal no tratado

Los agricultores acogidos a esta medida deberán de respetar los siguientes compromisos:

- Utilizar para la sementera exclusivamente semillas que no contengan productos fitosanitarios (semillas no blindadas ni tratadas)
- Preparar el terreno correctamente para el buen desarrollo de las plantas de leguminosa
- Comunicar la fecha de siembra al responsable del Programa de Medidas Agroambientales al menos 5 días antes de realizarla.
- No realizar tratamientos fitosanitarios sobre la parcela acogida durante el periodo de duración del compromiso (desde su siembra en el mes de octubre, hasta su retirada a partir del 10 de julio)
- La siembra se realizará en las fechas habituales para el cereal, no entrando a realizar ningún tipo de labor ni ninguna práctica, que contribuya a espantar a la fauna de las parcelas acogidas hasta el final del compromiso que será el 10 de julio.
- Los agricultores acogidos a esta medida estarán obligados a comprometer estas mismas parcelas en el siguiente ciclo agrícola, a la medida 1 "mejora y mantenimiento del barbecho tradicional", o bien, por segundo año repetir los compromisos de la medida número 5

Compromisos generales para todas las medidas

Todas las parcelas acogidas al programa de medidas agroambientales, independientemente de la medida a la que estén acogidos, deberán respetar unos compromisos de carácter general para todas ellas.

1. No utilizar productos fitosanitarios sobre la parcela acogida.
2. No utilizar semillas tratadas o blindadas para la sementera.
3. No realizar quema de rastrojos.
4. No permitir el paso y pastoreo de ganado.
5. No se permite el uso ni el vertido de compost o de lodos de depuradora.

Todos los compromisos establecidos son de renovación anual, durante un periodo máximo de 5 años

La dosis mínima de semilla que se recomienda para la sementera es de 180 Kg//ha para el trigo, 170 Kg/ha para cebada, 190kg/ha para la veza y 120 kg/ha para otras leguminosas.

PRIMAS COMPENSATORIAS

Los agricultores que decidan acogerse a este Programa recibirán una serie de primas compensatorias, cuyas cantidades dependerán del tipo de medida que deseen aplicar (Tabla 1). Es aconsejable que los agricultores se acojan a las medidas de "Extensificación de la producción agraria" ofrecidas por las administraciones autonómicas, que tienen un carácter similar a las que ofrece el presente Programa, para así conseguir aumentar la cuantía de las primas recibidas.

Tabla 1. Primas compensatorias según el tipo de medida

		Prima €/Ha/año
Medida 1	Mejora y mantenimiento del barbecho tradicional	138
Medida 2	Barbecho semillado con leguminosas	425
Medida 3	Retirada de la producción de tierras durante 4 años	287
Medida 4	Rotación de cultivos trigo-girasol	525
Medida 5	Cultivo de cereal no tratado	400

CONDICIONES DE CONCESION DE LAS AYUDAS

Para que las superficies puedan acogerse al programa de medidas agroambientales deben de cumplir los siguientes requisitos:

Estar incluida dentro de alguno de los 7 polígonos descritos como zonas de actuación del programa, estar situada en un área que por sus características sea susceptible de ser ocupada por comunidades de aves esteparias y alejada más de 1000 metros de núcleos de población, así como de carreteras asfaltadas, edificaciones habitadas, o cualquier otra obra o lugar en que la actividad humana pueda causar molestias frecuentes a las aves.

DOCUMENTACIÓN A PARESENTAR POR LOS SOLICITANTES

Los solicitantes de las ayudas, una vez que hayan recibido en su domicilio una carta con el documento de aceptación de las parcelas solicitadas, deben de presentar la siguiente documentación:

1. Factura que acredite la compra de semillas que no contengan productos fitosanitarios. En el caso de que la semilla sea propiedad del agricultor, deberá de presentar una declaración jurada de no haber realizado ningún tratamiento a la semilla destinada a la siembra de parcelas acogidas al programa.

2. En el caso de los agricultores que deseen acogerse a la medida 3, retirada de producción de la tierra durante un periodo máximo de 5 años, deberán de entregar una copia de una declaración o comprobante, que verifique el uso de la tierra a labores agrícolas durante los últimos 3 años.

3. Fotocopia del DNI, N.I.F ó C.I.F.

4. Impreso de aceptación de parcelas incluidas en el programa, debidamente rellenado y firmado.

PAGO DE LAS AYUDAS

Transcurrido el año agrícola y verificados los controles correspondientes, se enviará por correo a cada agricultor una factura con la cantidad correspondiente a la ayuda. Una vez firmada por el titular, éste deberá enviarla al Museo Nacional de Ciencias Naturales, tras lo cual se procederá a tramitar el pago de las ayudas a los agricultores con la mayor brevedad posible. Estos pagos serán efectuados en un pago único y por transferencia bancaria.

CONTROLES Y SANCIONES POR INCUMPLIMIENTO DEL CONTRATRO

El control por parte de la empresa sobre el no uso de productos fitosanitarios, abonos, calendario de labores y mantenimiento de barbechos y rastrojeras se realizará "in situ" sobre el 100% de las parcelas acogidas al programa y será realizada numerosas veces a lo largo del ciclo agrícola.

Cuando a través de los controles efectuados se comprueben anomalías en el cumplimiento de las condiciones y compromisos suscritos en alguna de sus partes, la empresa, en función de las circunstancias que concurran, podrá reducir las primas por unidad de superficie durante el ciclo agrícola en transición, si las condiciones del programa no han podido cumplirse en su totalidad, u optar por expulsar al propietario del programa en el caso de que las condiciones y objetivos señalados en el programa dejasen de cumplirse en su totalidad.

En el caso de incumplimiento de los compromisos establecidos en las medidas, se aplicarán las siguientes sanciones:

1. Cuando se compruebe una variación en las normas de cultivo establecidas que afecte entre el 5 y el 10 por ciento del total de la superficie de parcelas acogidas, se procederá a reducir la prima total a percibir en dicho año por el agricultor, calculada de acuerdo con las superficies reales, en un 20 por ciento.

2. Cuando la variación de la alternativa afecte entre el 10% y el 20% de la superficie, o aun siendo inferior implique una disminución de la superficie de leguminosa, se procederá a una disminución de la prima total a percibir en dicho año en un 50 por ciento.

3. Cuando la variación supere el 20%, se procederá a anular la prima y a rescindir el contrato

4. Cuando se compruebe reiteración durante más de un año en el incumplimiento de alguno de los compromisos establecidos por el programa, se procederá a la expulsión definitiva del propietario de las tierras del programa de medidas agroambientales

FINANCIACIÓN

Henarsa S.A., concesionaria del Estado para la construcción y explotación de las nuevas autopistas M-50 (tramo M-607 / N-IV, subtramo N-I / N-II) y R-2, a través del Proyecto de medidas preventivas, correctoras y compensatorias de la afección de las mencionadas infraestructuras a la población de avutardas y otras aves esteparias de la IBA "Talamanca-Camarma", y al LIC "Cuenca de los ríos Jarama y Henares", elaborado por el Museo Nacional de Ciencias Naturales, financiará en su totalidad las medidas que componen este Programa.

Situación de las siete zonas donde se aplicará el Programa:

1. Ribatejada-Valdeolmos-Valdetorres de Jarama-Talamanca de Jarama- Valdepiélagos

2. Valdetorres de Jarama-Fuente el Saz

3. Camarma de Esteruelas-Daganzo de Arriba-Alcalá de Henares-Fresno de Torote-

Ribatejada

4. Cobeña-Paracuellos de Jarama

5. Ajalvir-Daganzo de Arriba

6. Camarma de Esteruelas-Meco

7. Cabanillas del Campo-Quer-Villanueva de la Torre