Multipurpose use and restoration of wetlands in semiarid Mediterranean catchments degraded by intensive agricultural use

Utilización multipropósito y restauración de humedales en cuencas Mediterráneas semiáridas degradadas por el uso agrícola intensivo

Memoria presentada por **D. David Moreno Mateos** para optar al grado de Doctor por la Universidad de Alcalá de Henares.

Febrero de 2008

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Hacen constar

Que el trabajo descrito en la presente memoria titulado **"Multipurpose use and restoration of wetlands in semiarid Mediterranean catchments degraded by intensive agricultural use"**, ha sido realizado bajo su dirección por D. David Moreno Mateos en el Instituto Pirenaico de Ecología-CSIC, Jaca, y reúne todos los requisitos necesarios para su aprobación como Tesis Doctoral.

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Hace constar

Que el trabajo descrito en la presente memoria titulado **"Multipurpose use and restoration of wetlands in semiarid Mediterranean catchments degraded by intensive agricultural use"**, ha sido realizado por D. David Moreno Mateos bajo la dirección de los Drs. César Pedrocchi Renualt y Francisco A. Comín Sebastián en el Instituto Pirenaico de Ecología-CSIC, Jaca, y reune todos los requisitos necesarios para su aprobación como Tesis Doctoral.

Alcalá de Henares, 27 de febrero de2008

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A Pay y Nicolás,

por hacerme feliz

Agradecimientos/acknowledgements

Es maravilloso el momento en el que un hombre es capaz de tomar con pocas dudas la decisión correcta. Esto podría ser una definición más o menos acertada de la madurez humana. Este es un hecho cada vez más frecuente en mi vida y que sucedió cuando tomé la decisión de "probar con la investigación" y que sucedió con más éxito cuando le pedí a Pay que se casara conmigo y que volvió a suceder con aún más éxito, es difícil pero no imposible, cuando decidimos tener descendencia. Pero es poco probable que tomando una decisión uno solo esta llegue a alcanzar un elevado grado de éxito, para que esto suceda he comprobado que la presencia de otros es imprescindible.

Desde que "probé con la investigación" me he dado cuenta que este es mi camino, como ha ocurrido en numerosas ocasiones en mi vida el camino no es desde luego el más sencillo, más bien al contrario, es un camino profesional de extrema dificultad. Pero esto no tiene por que ser un problema si estás en el camino correcto. La posibilidad de dedicar todo mi esfuerzo laboral a aprender, crear, imaginar y proponer alternativas para mejorar la andadura del hombre por el mundo y tener una somera idea sobre el funcionamiento de algunas de sus partes merece el esfuerzo. Esto no quita que cuestione algunos de los planteamientos de la investigación, en especial en nuestro país.

Una vez que me he visto metido en el meollo de la investigación en ecología me he dado cuenta que realmente nunca ha sido "probar", sino que yo mismo me he ido dirigiendo en esa dirección, negando repetidamente atractivas alternativas que iban surgiendo por el camino. Todo encaja perfectamente cuando pienso en lo que me gustaba estar en el campo de pequeño y ver los bichos, de ahí mi temprana afición a la observación de aves (¡desde los 10 años!), mi pasión por ir al campo, que se ha ido desarrollando enormemente durante toda mi vida. Hace mucho tiempo, diría toda la vida, que me sentía atraído por saber como funcionaba todo eso. Todo es demasiado, pero conocer una ínfima parte de la maquinaria ha sido simplemente lo que yo he buscado, es decir, el camino correcto.

En primer lugar, quiero agradecer a César Pedrocchi y Paco Comín por darme aquella oportunidad de "probar con la investigación", gracias a la que, por fin, di con el camino correcto. Ha sido un camino largo que afortunadamente se ha ido estrechando y aportando cada vez más. Sin su apoyo, dedicación e interés esto no habría salido así. A César quiera agradecerle el acercarse como persona, su apoyo incondicional en muchas de mis peticiones y el interés por seguir peleando por buscar un mundo mejor a pesar de todo, ¡y a cuatro años de jubilarse!. A Paco quiero agradecerle fundamentalmente el confiar en mí, además de crear una relación profesional madura, próxima y en crecimiento. A los dos espero que tras la distancia volvamos a juntarnos para construir un equipo fuerte que crezca y siga soñando. ¡¡¡Muchas gracias a los dos!!!. En segundo lugar, agradezco enormemente la ayuda y confianza depositada en mí por mi tutor y amigo, José María Rey Benayas. También, quiero agradecerle la infinidad de pequeños favores que me ha hecho, sus consejos estadísticos, su ánimo constante y su relación de amigo, ¡¡¡seguimos adelante!!!

En tercer lugar, quiero agradecer a otras personas del IPE su interés y dedicación a las muestras que llevaba, en especial quiero agradecer a Mercedes García el aguantar mis permanentes presiones para que saliesen los resultados y ayudarme en momentos críticos, ¡gracias por enfadarte tan poco!. También agradezco enormemente a Santiago Pérez su ayuda inestimable en el laboratorio de Jaca, sus charlas infinitas sobre el funcionamiento de las cosas y su auténtico ánimo, ¡gracias Santi!. También, quiero agradecer la colaboración y ayuda de Ramón Galindo por sacar a tiempo la infinidad de muestras del suelo, en general, a todo el personal investigador, auxiliar y de administración que de una u otra manera han ayudado en este proceso.

Bajando (o subiendo) al plano becaril/amistoso, en primer lugar agradezco mucho que Alvaro Cabezas (alvarito) haya estado ahí desde el principio, ¡¡¡muchas gracias y ánimo que te queda poco!!!. Y en Jaca ha sido fundamental que Guillermo Bueno (Guille) estuviera ahí para hablar de la vida, la investigación y las parejas, y comer en su casa tantas veces, y ser una estupenda persona, ¡¡¡genial guille y ánimo que también queda poco!!!. También quiero recordar a otras personas estupendas que están o han estado en Jaca, Sergio Couto, Juan Cervantes, Emilia Martín, Gonzalo Matínez, Marcos Regueira y Darío Abadías. Y a otras personas estupendas que están o han estado en Zaragoza como Sara Palacio (gracias por tu superayuda), Cecilia Ciancarelli, Belinda Gallardo y María González. Fuera del IPE agradezco su gran ayuda y amistad a Almudena Rodríguez, ¡gracias y ánimo!.

Pero hay más gente sin la que esto no habría sido posible, en primer lugar, muy por encima de todo esto están mis dos amores, Pay y Nicolás. Sin Pay sencillamente nada sería posible, su amor verdadero, su confianza en mí, su apoyo constante y su fuerza para ser cada vez mejor me hacen cada día ser más feliz, y para rematar, ha venido Nicolás el fruto perfecto de nuestro amor, que es en sí el amor más puro que jamás he conocido, ¡¡¡sois mi vida!!!. Mis padres (Antonio y Delia) han tenido un papel crucial también en este desarrollo de los hechos, su confianza, su amor incondicional y su apoyo constante han permitido que todo este tren llegue a un maravilloso destino, ¡¡¡gracias por creer en mi!!!. A mis hermanos, Miguel (y Ester), por su amor, y Jaime, por su amor y por buscar la alternativa de ser un hombre de verdad. También quiero agradecer a algunos de mis amigos que llevo siempre en el corazón por su apoyo, amistad y confianza en este proceso, Fernando Santos y Belén Zubieta, Carlos Ruiz y Lise Pomarede, Guille Olague y Silvia Rull y Ana García. Quiero recordar a mis amigos de Huesca que me han ayudado y apoyado constantemente, Dani y Ana, Rober y Lore, Ruben y Lurdes y José Ángel y Gema (a estos últimos gracias especiales por ayudar notablemente con la forma de la tesis). Quiero agradecer profundamente a Adolfo Herrero por ser así y por enseñarme a ser un hombre, a mis suegros Reyes y Helmi por creer en su familia (incluido yo, siempre estaremos con vosotros en ese camino), y a Josele (Boletas) por su confianza, ayuda y amistad.

Finally, I would like to remember those wonderful moments in the smoke sauna and acknowledge the useful scientific advise, the friendship and the ability to create wonderful moments among the people to one Estonian guy, ¡¡¡many thanks Ülo Mander!!!.

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Summary

An increasing interest exists on restoring wetlands worldwide because of the valuable functions and services they provide. This study aims to demonstrate that using wetlands with multiple objectives to restore semi-arid areas degraded by intensive agricultural irrigation is possible. Besides, it aims to provide a set of guidelines for wetland design and construction which ensure their landscape integration and efficient functioning at large scale. To do this, wetlands have been studied in the Flumen river catchment (Monegros, Aragon, NE Spain) integrating four perspectives: water, soil, landscape and biodiversity. The items studied experimentally were: the efficiency of existing and constructed wetlands to improve the quality of the agricultural run-off, the characteristics of soils in wetlands flooded with agricultural run-off faced to conventional agricultural fields, the role of wetlands in the landscape using geographic information systems, the seasonal structure and dynamics of bird communities existing in them.

The quality of agricultural water run-off improves using wetlands for its treatment. Almost all the nitrate passing through wetlands of adequate size is eliminated. The salt content of the soil is significantly reduced in four years when water flows permanently. Organic matter accumulation is slow in the wetlands but it improves the quality of degraded agricultural soils. Water quality, biodiversity and soil improvement are favoured by a scattered spatial distribution of restored wetlands in an agricultural catchment. Wetlands restoration planning must be implemented at basin scale, beginning to restore according to the needs of small catchments. Wetlands can fulfil several objectives simultaneously if restrictions for every objective and local limitations of the study area were deeply known.

The results of this study provide tools to integrate the restoration and creation of wetlands with separated or multiple objectives in the socio-economical development of semiarid areas extensively used for intensive irrigated agriculture.

Resumen

El interés en restaurar humedales aumenta de forma constante en todo el mundo por las valiosas funciones y servicios que cumplen. Este trabajo pretende demostrar que es posible la utilización de humedales con múltiples objetivos para la restauración de zonas agrícolas degradadas por riego agrícola intensivo. También pretende aportar una serie de directrices para el diseño y construcción de humedales de forma que queden integrados en el paisaje y sean funcionales a escala territorial. Para ello, se han estudiado humedales en la cuenca del río Flumen (Monegros, Aragón, NE España) integrando cuatro perspectivas: el agua, el suelo, el paisaje y la biodiversidad. Se ha estudiado la capacidad de humedales existentes y construidos para mejorar la calidad del agua excedente de riego agrícola, las características del suelo en humedales bañados con agua excedente de riego frente a la de campos agrícolas, se han determinado el papel de los humedales en el paisaje mediante sistemas de información geográfica y se ha estudiado la estructura y dinámica estacional de comunidades de aves presentes en ellos.

La calidad del agua excedente del riego agrícola mejora, se elimina la práctica totalidad del nitrato, a su paso por los humedales cuando estos tienen el tamaño adecuado. El paso continuo de agua reduce la salinidad del suelo en cuatro años. La acumulación de materia orgánica es lenta pero mejora la calidad de los suelos agrícolas degradados. Una distribución espacial dispersa y numerosa de humedales restaurados en una cuenca agrícola favorece la mejora de la calidad del agua, la biodiversidad de aves y la recuperación de suelos. La planificación de proyectos de restauración de humedales debe realizarse a escala de grandes cuencas, empezando a restaurar según las necesidades de cuencas pequeñas. Los humedales pueden cumplir varios objetivos simultáneamente siempre que se conozcan en profundidad las necesidades de cada objetivo y las limitaciones de la zona objeto de restauración.

Los resultados de este trabajo proporcionan herramientas para la integración de los humedales con objetivos aislados o integrados diversos

de mejora de la calidad del agua y suelo, la biodiversidad y el paisaje, en el desarrollo socio-económico de zonas semiáridas con intensa y extensa actividad agrícola de regadío.

Chapter 1

General introduction/Introducción general

Degradación y recuperación de ecosistemas en zonas agrícolas semiáridas

Regiones semiáridas

Las regiones semiáridas se caracterizan normalmente por su baja productividad agrícola debido a la escasez de los recursos hídricos inherentes a su climatología. En estos ambientes es frecuente que la cobertura vegetal sea escasa, muchas veces consecuencia de una excesiva carga ganadera, y se limita a formaciones herbáceas y arbustivas, lo que facilita el arrastre de sedimentos y los procesos erosivos. También es frecuente, como consecuencia de las reducidas precipitaciones, que las concentraciones de sales en el suelo sean elevadas, lo que dificulta aún más la supervivencia de las especies. Por otro lado, la humedad ambiental es reducida y la insolación elevada (Pedrocchi, 1998b). Bajo estas condiciones (escasa producción biológica, arrastre de sedimentos, sequedad ambiental y, comúnmente, salinidad) la acumulación de la materia orgánica en el suelo es muy baja. Además, la actividad microbiana actúa más lentamente y la materia orgánica se descompone casi exclusivamente por fotodegradación (Austin and Vivanco, 2006). Esta situación no permite el asentamiento de ecosistemas altamente productivos y, en general, la diversidad y abundancia de especies de las comunidades asentadas en él es reducida. Como contrapartida, en muchos casos estas comunidades tienen un elevado valor ecológico por su especificidad (Margalef, 1974).

Degradación de los ecosistemas semiáridos

Los asentamientos humanos en las zonas semiáridas suelen estar adaptados a sus condiciones y, por consiguiente, se establecen en números reducidos. La explotación del suelo en estas zonas se desarrolla normalmente en dos niveles, por un lado se cultivan en secano laderas suaves y zonas llanas con varios años de descanso, y por otro lado, se trabajan en regadío y de forma permanente las zonas de mayor productividad, como son vaguadas y fondos de valle. Incluso estas explotaciones de baja intensidad ejercen un fuerte efecto sobre el frágil ecosistema favoreciendo la erosión y la perdida de diversidad.

Para contrarrestar esta situación de despoblación frecuente en zonas semiáridas, se han llevado a cabo por todo el mundo grandes obras de transformación al regadío habitualmente con consecuencias impredecibles y, en muchos casos, catastróficas. Ejemplos de esta situación han sido la desecación del Mar de Aral y las llanuras inundables de África Central por la extracción de agua, o el retroceso del delta del Nilo por la disminución de los sedimentos ocasionada por la presa de Aswan (Lemly et al., 1993; Galbrait et al., 2005). La transformación de un territorio al regadío conlleva tres grandes transformaciones. En primer lugar, la traída de agua llevada a cabo mediante presas en los ríos y canalizaciones, o bien, mediante bombeos de aguas superficiales a menor cota o subterráneas. En segundo lugar, una adaptación de los campos de cultivo al riego, que va desde la red de distribución hasta el riego de cada campo, esto puede necesitar una transformación total de la topografía del terreno para su nivelación, si el riego es por inundación, o una suavización del terreno y la instalación de medios de aspersión o goteo. En tercer lugar, la evacuación del agua residual agrícola mediante una amplia red de drenaje cuyo objetivo es sacar el agua lo más rápidamente posible de la zona agrícola a la red hídrica general.

La fragmentación de los hábitats es uno de los impactos más destacables de la propia transformación. Las manchas originales de diferentes comunidades vegetales suelen estar reducidas a mínimos que no aseguran la funcionalidad del ecosistema. En consecuencia, se produce una pérdida de la biodiversidad y el incremento de unas especies (animales y vegetales) más oportunistas en detrimento de las originales (Lemly et al., 2000). La eliminación en su mayor parte del paisaje original da paso a grandes extensiones del territorio cubiertas casi exclusivamente por campos de cultivo, reduciendo la heterogeneidad paisajística drásticamente (Foto 1; Comín et al., 2001). En este caso la escasa diversidad es aportada por reducidos parches de vegetación natural que quedan en zonas no aptas para la agricultura.

Una vez puesto en marcha el regadío (Fotos 2 y 3) aparecen nuevos impactos negativos para los ecosistemas locales. El incremento de los procesos erosivos aparece como consecuencia de la descompactación del suelo por los movimientos de tierra y del aumento de la escorrentía superficial (Foto 4). Uno de los impactos más extendidos y estudiados en todo el mundo es el incremento de nutrientes (especialmente nitrógeno y fósforo) y compuestos orgánicos en el agua por la aplicación de fertilizantes y plaguicidas. Este problema tiene una repercusión muy importante por los graves problemas de eutrofización que sufren los ecosistemas acuáticos (Galbrait et al., 2005; Zedler and Kercher, 2005). Problemas agudos de eutrofización de grandes masas de agua dulce como el lago Taihu han puesto en peligro la salud humana en ciudades como Shangai (Guo, 2007). Estos problemas también pueden aparecer en aguas marinas cercanas a las desembocaduras de grandes ríos como el Mississippi. En el Golfo de México existe actualmente una zona de hipoxia de unos 20,000 km^2 que ha afectado a las comunidades marinas aunque sin efectos actualmente notables sobre la productividad pesquera de la zona (Mitsch et al., 2001).

La degradación del suelo puede ser un impacto muy importante en algunas zonas transformadas al regadío (Foto 5). Numerosas zonas en todo el planeta tienen una naturaleza más o menos salina que bajo condiciones de aridez natural no presenta problemas graves. Una vez que se incrementa el aporte de agua estas sales pueden ser liberadas progresiva y continuamente a la red hídrica, como sucede en grandes extensiones de Australia (Rengasamy, 2006) y en el valle del Ebro , España (Causapé et al., 2004). En este último caso, la eliminación de los horizontes superficiales en amplias extensiones de los suelos originales ha dejado al descubierto la roca madre (lutita) de naturaleza evaporítica y con un gran contenido en sales (Fig. 1; Rodríguez-Ochoa et al., 1998).

En el caso de Los Monegros (NE de España; Foto 6) en la cuenca del Ebro todos los impactos mencionados han ocurrido en las extensas zonas que han sido transformadas en regadío (110,000 ha). Se trata en la actualidad de una zona degradada con baja diversidad, cuyas aguas tienen una gran carga de nutrientes, con suelos con graves problemas de salinización-sodificación y con un paisaje altamente homogéneo. Esta situación se ve agravada por el abandono institucional causado probablemente por la baja densidad de población de la zona (~7 habitantes/km^2) y por el interés económico menor que tiene respecto a zonas donde el turismo o la industria tienen más presencia. La transformación al regadío se realizó a partir de los años 40 para facilitar el asentamiento de población en zonas despobladas procedente de otras regiones del país a través del antiguo Instituto Nacional de Colonización. Aunque ya no viene más población, la transformación al regadío sigue realizándose aún hoy en día, cada año varios miles de has de secano son transformadas a regadío (12.000 ha entre 2004 y 2007). Ni las nuevas transformaciones ni las antiguas toman ninguna medida que asegure la reducción de la contaminación del agua, que evite la degradación de los suelos y del paisaje, ni que fomente la biodiversidad.

Figura 1. Alteración del suelo producida tras la transformación en regadío y la aparición posterior de nuevos horizontes orgánicos como consecuencia de la presencia de humedales.

Aparición de nuevos humedales en Los Monegros

El incremento de la escorrentía superficial ha producido importantes cambios en las comunidades vegetales. Han aparecido numerosas zonas donde existe un excedente de riego acumulado que ha permitido el asentamiento de comunidades de especies acuáticas (Fig. 2) principalmente dominadas por el carrizo (*Phragmites australis* (Cav.) Trin. ex Steud.) y acompañados por algunas especies de juncos (*Scirpus* sp. y *Carex* sp.) y eneas (*Typha latifolia* L.). Estos nuevos humedales se pueden clasificar en dos tipos fundamentales, aquellos que aparecen en laderas como consecuencia de infiltración de un exceso de agua de riego (Fig. 2; Foto 7), donde no es posible una acumulación de agua en el fondo, y aquellos que se encuentran en fondos de valle más o menos planos (Foto 8), donde si es posible la acumulación de agua en el fondo.

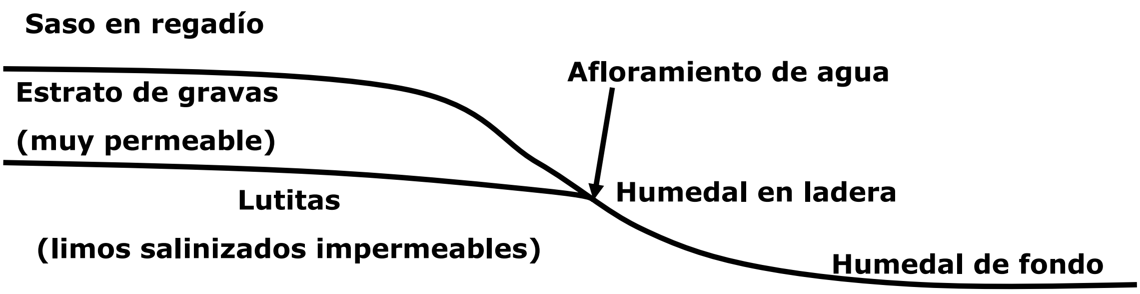

Figura 2. Zonas de aparición de humedales espontáneos como consecuencia del exceso de agua tras la transformación en regadío. Saso es el nombre local que reciben las mesetas.

Estos humedales espontáneos tienen una gran importancia en los procesos de mantenimiento de la biodiversidad y la calidad del agua (véase Capítulos 1 y 4), pero su funcionalidad es aún baja respecto al óptimo que se podría alcanzar para lograr estos dos objetivos. La aparición de humedales ha permitido en muchos casos la aparición de un pequeño horizonte orgánico que permite empezar a recuperar la fertilidad perdida del suelo (Fig. 1). El estudio de estos humedales ha sido clave para conocer su estado y poder proponer medidas para su conservación y mejora.

Utilizando humedales para restaurar ecosistemas

Una vez se plantea la necesidad de recuperar zonas degradadas existen varias alternativas de restauración. La más sencilla sería no actuar y dejar que las fuerzas de la naturaleza actúen solas haciendo el trabajo (Urbanska et al., 1997). Si se decide actuar, encontramos múltiples alternativas, siempre teniendo en cuenta que en casos de restauración a gran escala la intervención humana debe ser la menor posible que asegure el éxito de la restauración y que favorezca la rehabilitación de los procesos que regulan los ecosistemas. Debemos también considerar que el éxito de la restauración, es habitualmente complejo de determinar (Ruiz-Jaen and Aide, 2005). Tendríamos dos alternativas fundamentales, recuperar el ecosistema original, si se conoce cual fue detalladamente, o crear un nuevo ecosistema con condiciones similares o diferentes respecto al ecosistema original. En este punto entra en juego el objetivo que persiga la restauración. Si se persigue recuperar la cubierta vegetal, por ejemplo, para frenar la erosión, entonces la restauración se podría acercar al ecosistema original. Pero si la coyuntura territorial necesita alcanzar otros objetivos con ese ecosistema restaurado, por ejemplo, mejorar el paisaje, reducir la contaminación del agua y frenar la erosión, entonces se puede restaurar con una perspectiva multipropósito aumentando las funciones ofrecidas por el ecosistema restaurado.

Entonces, de entre las alternativas válidas para restaurar un ecosistema degradado en este estudio se escoge utilizar un ecosistema diferente del original por la necesidad territorial de cumplir varios objetivos de restauración. En este punto, nos podemos plantear que entonces estamos en la modalidad de restauración conocida como creación en la literatura *ad hoc* puesto que creamos algo que nunca antes había existido. En este sentido, dada la enorme cantidad de ecosistemas, concretamente humedales, que han desaparecido en todo el mundo (aprox. un 50%; Zedler and Kercher, 2005) se trata de recuperar superficie original y, sobre todo, las funciones de los humedales, aunque no sea en el mismo lugar.

Tabla 1. Objetivos más comunes perseguidos con la restauración de humedales y algunos de los estudios más relevantes. Se debe añadir el objetivo recreativo, muy común en proyectos de restauración que no se ha incluido por la inexistencia de trabajos científicos específicos.

Objetivo de restauración	**Caso**
Mejora de la calidad del agua	(Hammer, 1989; Hammer, 1992; Kadlec and Knight, 1996; Romero et al., 1999; Woltemade, 2000; Koskiaho et al., 2003; Mitsch et al., 2005a)
Conservación de la biodiversidad	(Worrall et al., 1997; Comín et al., 2001; Hansson et al., 2005; Takekawa et al., 2006)
Control de inundaciones	(Hey and Philippi, 1995; Zedler, 2003; Cox et al., 2006)
Secuestro de carbono	(Whiting and Chanton, 2001; Mitra et al., 2005; Zedler and Kercher, 2005)
Conservación de la pesca	(Koonce et al., 1996; Kaly and Jones, 1998; Richardson and Hussain, 2006)

La restauración de humedales se está convirtiendo en una práctica común en muchos países del mundo (Zedler, 2000), y cada año se emprenden más proyectos de restauración, especialmente en los países desarrollados. Existen numerosos objetivos para la restauración de humedales (Tabla 1), algunos muy extendidos y otros de reciente aparición (véase Capítulo 8) y se han empezado a estudiar las posibilidades de restaurar humedales con una perspectiva multipropósito (Comín et al., 2001; Hansson et al., 2005).

A día de hoy se tiene una pequeña base de conocimiento científico que permite que los proyectos de restauración tengan un mayor grado de éxito. Esto ha sido posible gracias a los numerosos estudios que han permitido conocer en mayor detalle el funcionamiento ecológico de los humedales (Mitsch and Gosselink, 2000a; Zedler, 2003; Zedler, 2005). Ya existen extensos manuales sobre la restauración de múltiples tipos de humedales como compendios de toda la información existente (Eades et al., 2005). Pero aún existen enormes carencias en el conocimiento de los humedales y su restauración. Existen numerosos ejemplos de proyectos de restauración donde no se siguen la trayectorias planteadas inicialmente y no se han cumplido los objetivos de las restauraciones (Zedler and Callaway, 1999). Por ello, la profundización en su conocimiento permitirá que en el futuro la restauración de humedales tenga más éxito y pueda ofrecer un abanico de servicios a las regiones sobre las que se asiente.

Hipótesis general y objetivos

La **hipótesis general** de la tesis plantea que "**es posible recuperar ecosistemas degradados por el uso agrícola intensivo mediante la utilización de humedales restaurados**". Como conclusión de todo el trabajo se demuestra que esto es posible y se plantean medidas para conseguirlo.

Así, el **objetivo general** de la tesis es conocer en que grado se recuperan los ecosistemas degradados por el uso agrícola intensivo con la utilización de humedales restaurados en una zona Mediterránea semiárida y proponer medidas de diseño y gestión de los mismos para optimizar su funcionalidad. Este objetivo general pude desglosarse en cinco **objetivos específicos**:

1. Estimar los cambios en los parámetros del agua residual agrícola al pasar a través de humedales de flujo superficial construidos sobre suelos naturales en antiguos campos agrícolas de regadío.
2. Evaluar el efecto que ejerce la presencia de humedales permanente e intermitentemente inundados sobre las principales características de suelos agrícolas degradados.
3. Averiguar las relaciones existentes entre los parámetros del paisaje, las características de los humedales crecidos espontáneamente tras la

transformación al regadío y la calidad del agua en los ríos. Y demostrar un método de análisis del paisaje que pueda ser utilizado para estimar la idoneidad de una cuenca para la construcción o restauración de humedales.

4. Definir las características de los humedales crecidos espontáneamente tras la transformación al regadío que condicionan la estructura de las comunidades de aves.
5. Como síntesis de los demás objetivos específicos se pretende además aportar una orientación útil sobre como restaurar o crear humedales con perspectivas multipropósito en cuencas agrícolas degradadas.

La región de Los Monegros

Monegros es una amplia región de 270.000 ha repartida entre las provincias de Huesca y Zaragoza (Comunidad Autónoma de Aragón) que se encuentra dentro de la cuenca del río Ebro, NE de España. La mayor parte de esta región se encuentra actualmente dentro de los límites administrativos de la Comarca de Los Monegros (Fig. 3). Es una zona poco poblada (~7 habitantes·km^2) cuya ocupación mayoritaria es la agricultura. Aproximadamente un tercio de la superficie esta transformada al regadío. Más de un 80% de la superficie regada está cultivada con maíz y alfalfa. Otros cultivos en regadío extendidos por orden de importancia son cereal, arroz, hierba, girasol y, ocasionalmente, hortícolas o frutales. También hay numerosas explotaciones de ovino y vacuno, y en los últimos años ha habido un explosión en el número de granjas porcinas alcanzándose 1.000.000 de porcino en la zona en 2007. Esto está generando graves problemas de contaminación incontrolada de los cursos de agua.

En esta región se encuentra el río Flúmen, un tributario del Cinca cuyos tramos medio y bajo son el área de estudio. El clima es semiárido y Mediterráneo continental. La temperatura media anual es de 14.5ºC y la precipitación media anual de 400 mm (Pedrocchi, 1998b), pero presenta una elevada variabilidad interanual (Comín and Williams, 1993).

Los suelos están principalmente dominados por una estructura Terciaria que consiste en lutitas con diferentes grados de salinidad (conductividad entre 1 y 10 mS cm^{-1}) con estratos insertados de arenisca. Localmente, esta estructura está cubierta por capas de grava de 2-4 m depositadas como glacis procedentes de las sierras cercanas, principalmente la sierra de Alcubierre. La vegetación pseudo-esteparia original, que estaba dominada por romero (*Rosmarinus officinalis*), tomillo (*Thymus* sp.), algunas halófitas (*Salicornia* sp.) y herbáceas perennes (*Lygeum spartium*, *Brachypodium retusum*), ha cambiado a un paisaje dominado por especies halófitas (*Atriplex* sp. and *Suaeda vera*) en las zonas secas elevadas y helófitas (*Phragmites australis*, *Typha latifolia*, *Scirpus holoschoenus*, *S. maritimus, Carex divisa* y *Tamarix africana*) en las zonas húmedas de las partes bajas de los valles.

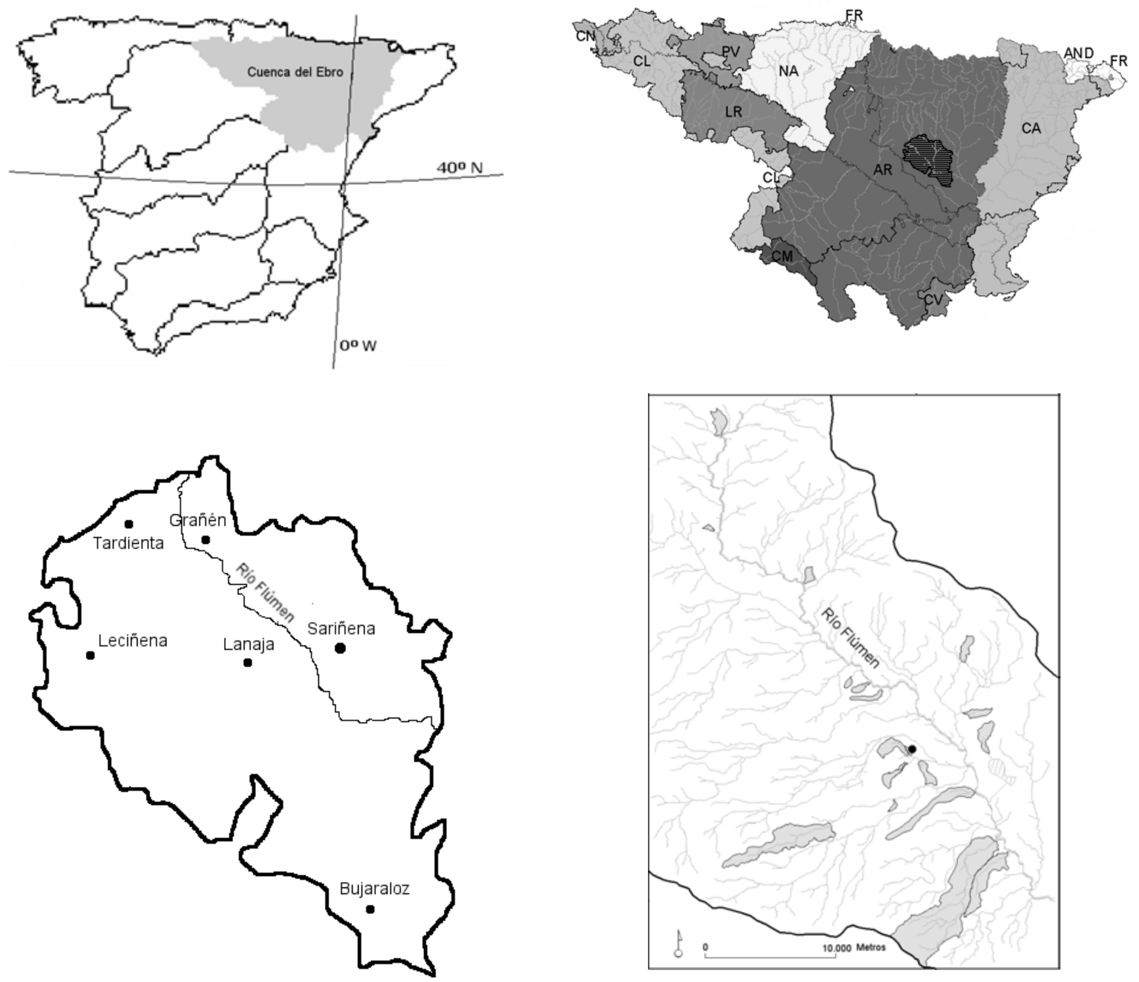

Figura 3. Cuencas hidrográficas más importantes de la península ibérica (arriba izquierda). Distribución administrativa de la cuenca del Ebro, sombreado en negro la Comarca de los Monegros (arriba derecha). Comarca de Los Monegros con los principales núcleos de población (>1.000 habitantes; abajo izquierda). Cuenca del Flúmen con las cuencas estudiadas en sombreado, el punto negro indica la ubicación de los humedales experimentales (abajo derecha).

Estructura de la tesis

Para demostrar la hipótesis general y cumplir los cinco objetivos propuestos se construyeron una serie de humedales experimentales sobre suelos agrícolas abandonados (Fotos 9 y 10) y se estudiaron otros humedales ya existentes y localizados en cuencas agrícolas transformadas al regadío (Fotos 7 y 8). La tesis se ha estructurado en ocho capítulos además de la introducción. Seis dedicados a desarrollar los cuatro primeros objetivos de la tesis, uno más para cumplir el objetivo quinto a modo de revisión y síntesis. Se añade uno final de conclusiones generales.

Tras la introducción, en el **segundo capítulo** se hace una revisión inicial de los resultados obtenidos en los dos primeros años de funcionamiento de los humedales experimentales en cuanto a su capacidad para retener nutrientes (nitrógeno y fósforo). Además, se hace una pequeña aproximación a la escala paisajística estudiando la influencia de los humedales existentes sobre la heterogeneidad del paisaje. En el **tercer capítulo** se valora la evolución de los humedales experimentales tras cuatro años de funcionamiento con sus capacidades para retener nutrientes (nitrógeno, fósforo y carbono) y sales, y como afecta el paso del agua residual agrícola a través de los humedales sobre otras variables relevantes para caracterizar la calidad del agua como son los sólidos, la alcalinidad, la turbidez y otros cationes como el calcio o el magnesio.

En el **cuarto capítulo** se evalúa el efecto que tienen la presencia de humedales sobre las características básicas del suelo (materia orgánica, salinidad y composición) con dos tratamientos, permanente e intermitentemente inundados. Esta distinción se hizo por el conocido efecto de las lutitas (margas salinizadas) en la liberación de sales cuando son inundadas y secadas.

En el **quinto capítulo** se estudian las características que debe tener un humedal para favorecer el establecimiento y conservación de comunidades de aves y, por lo tanto, incrementar la biodiversidad en zonas agrícolas degradadas.

En el **sexto capítulo** se salta de escala para estudiar el efecto de los parámetros paisajísticos tanto de las cuencas agrícolas como de los humedales existentes sobre la calidad del agua. Esta aproximación paisajística a la restauración de humedales permite ver que las actuaciones se deben plantear no sólo a nivel puntual sino también a escala territorial, preferiblemente de cuenca hidrográfica. En el **séptimo capítulo** se sigue a escala de paisaje para proponer un modelo simple que facilite la localización óptima de lugares donde restaurar humedales en cuencas agrícolas bajo criterios económicos, hidrológicos, geológicos y ecológicos.

El **octavo capítulo** revisa toda la información existente en los anteriores y la compara con otros estudios similares sobre restauración de humedales con diferentes perspectivas para evaluar la idoneidad de llevar a cabo restauraciones multipropósito o unipropósito y aportar un protocolo de actuación.

Finalmente, el **noveno capítulo** desarrolla una síntesis general de los capítulos anteriores, resumiendo las principales conclusiones obtenidas.

Esta estructura de la tesis por artículos ayuda a entender de forma independiente los resultados y conclusiones obtenidos en cada capítulo y sigue la secuencia lógica que ha seguido la tesis desde su planteamiento. Existen dos temas, la calidad del agua y el paisaje, sobre los que hay dos capítulos escritos de cada uno. Se han mantenido como capítulos independientes por que la cantidad de información obtenida en cada uno de ellos se consideró suficiente para que fuesen capítulos completos. De los siete capítulos centrales uno se encuentra publicado en *Ecological Engineering*, otros dos estás aceptados en *Ecological Engeneering* y *Applied Soil Ecoloy*, otros tres se encuentran enviados a revistas incluidas en el *Science Citation Index* y el último está en preparación para ser enviado. Por ello, la estructura de los capítulos se ha mantenido lo más parecida posible a los manuscritos aceptados o enviados a las revistas para su publicación. Esta estructura puede llevar a cierta repetición en la información aportada en la tesis, especialmente en los apartados de "material y métodos" o "zona de estudio", pero facilita la comprensión de los capítulos de forma totalmente independiente. Finalmente, la tesis ha sido escrita en ingles (capítulos centrales) y español (introducción y conclusiones generales). El inglés fue elegido por ser el idioma más utilizado en la ciencia y exigido en las publicaciones, y el español para mantener su importancia secundaria como idioma científico y acercar los resultados a hispanoparlantes que no hablen inglés.

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Fotografías

Foto 1. Foto aérea de una parte de la zona de estudio que muestra la transformación casi completa del territorio para el regadío. La elipse roja muestra la ubicación de los humedales experimentales.

Foto 2. Infraestructuras de la transformación al riego, red de distribución (acequias) y red de desagüe (azarbes).

Foto 3. Nivelación por terrazas habitual en las transformaciones con riego a manta.

Foto 4. Procesos erosivos y pérdida de la biodiversidad en terrazas sobresuelos salinizados

Foto 5. Pérdida de suelos y afloramiento de las lutitas (salinización).

Foto 6. Paisaje de la zona de estudio tras la transformación al riego. En primer plano riego por aspersión (terrenos suavizado), en el fondo riego a manta (terreno nivelado).

Foto 7. Humedal espontáneo aparecido por el afloramiento del exceso de agua en el regadío de la zona superior.

Foto 8. Humedal espontáneo aparecido tras la transformación al riego en el fondo de un pequeño valle agrícola.

Foto 9. Construcción de los humedales experimentales.

Foto 10. Humedales experimentales, se pueden apreciar las réplicas de los diferentes tamaños.

Chapter 2

Creating wetlands for the improvement of water quality and landscape restoration in semi-arid zones degraded by intensive agricultural use*

Abstract

Increasing interest on restoring wetlands with multipurpose approach is observed in degraded lands submitted to intensive human uses. This study evaluates the effectiveness of constructed and natural wetlands at removing nutrients from agricultural wastewater and their potential contribution to landscape heterogeneity in semiarid Monegros area, NE Spain. To achieve the first aim, wetland plots of differing sizes (50 m^2, 200 m^2, 800 m^2) were constructed upon fields abandoned four years earlier. Water has been sampled at the inflow and outflow of the plots for two years. Results show a 24-43% of total nitrogen removal and no clear trend for phosphorus in constructed wetlands. Slight effectiveness improvements took place in the second working year and in large size plots. For the second aim, a number of catchments with wetlands originated as consequence of irrigation were selected. These wetlands do not contribute significantly to improve the landscape diversity of agricultural catchments. Based on this experimental work, it is estimated that it should be necessary to restore wetlands in 3.25 and 5.60% of total watershed areas to remove most nitrogen from wastewater. Their restoration should be implemented also with the objective of increasing the landscape diversity of zones extensively transformed and homogenized by agricultural practices.

Resumen

Existe un creciente interés en restaurar humedales con una perspectiva multipropósito en zonas degradadas por un uso agrícola intensivo. Este estudio evalúa la efectividad de los humedales construidos y naturales en la eliminación de nutrientes de aguas residuales agrícolas y su potencial contribución al incremento de la heterogeneidad paisajística en la zona semiárida de Monegros, NE de España. Para conseguir el primer objetivo, se construyeron humedales de diferentes tamaños (50 m^2, 200 m^2, 800 m^2) sobre suelos agrícolas abandonados cuatro años antes. El agua fue muestreada a la entrada y la salida de los humedales durante dos años. Los

* Moreno, D., Pedrocchi,C., Comín, F.A., García, M. and Cabezas, A. 2007. *Ecological Engeneering* 30: 103-111

resultados mostraron una eliminación del 24-43% de nitrógeno total y una tendencia no clara para el fósforo en los humedales construidos. El segundo año de funcionamiento tuvo lugar un ligero incremento de la efectividad, al igual que en los humedales de mayor tamaño. Para lograr el segundo objetivo, se seleccionaron una serie de cuencas con humedales originados como consecuencia del regadío. Estos humedales no contribuyeron de forma significativa a la mejora de la diversidad paisajística de las cuencas agrícolas. Basándose en este trabajo experimental, se estimó que sería necesario restaurar humedales en el 3.25-5.60% del total de la superficie de la cuenca para eliminar la mayor parte del nitrógeno del agua residual. Su restauración debe ser llevada a cabo, además, con el objetivo de incrementar la diversidad del paisaje de zonas ampliamente transformadas y homogeneizadas por las prácticas agrícolas.

Introduction

Wetland restoration should integrate objectives across spatial scales in order to cover the range of natural interactions (Mitsch and Gosselink, 2000a; Zedler, 2000). Wetland construction is mostly focused on water quality improvement, although there is an increasing scientific interest for multipurpose approaches (Wheeler, 1995; Comín et al., 2001).

The capacity of constructed wetlands to remove nutrients from agricultural runoff has evolved mostly based on their nutrient removal efficiencies and hydraulic characteristics (Brix and Schierup, 1989; Hammer, 1992; Kadlec and Knight, 1996; Romero et al., 1999). New interest is arising now on restoring and creating wetlands to buffer non-point source pollution at watershed scale (Raisen and Mitchell, 1995; Mitsch and Gosselink, 2000a) because nutrients are responsible for eutrophication of natural aquatic ecosystems in river basins and coastal seas (Goldman and Horne, 1983).

Nutrient retention differs depending on factors such as nutrient concentration, seasonality, hydraulic loading, water-residence time, soil type, plant species and water chemistry. Results for nitrogen removal use to be as high as 99% of the inflow concentration, whereas for phosphorus are very variable (Zurayk et al., 1997; Mitsch and Gosselink, 2000a; Braskerud, 2002a; Braskerud, 2002b; Callaway et al., 2003; Kadlec, 2003; Koskiaho et al., 2003; Bruland and Richardson, 2004; Fink and Mitsch, 2004). Increasing residence time favors nutrients removal (Kadlec and Knight, 1996). However, it is well known that reaching effective nutrient removal may take a few growing seasons because of lack of well developed below and above-ground plant-microbial interactions (Mitsch and Jørgensen, 2004). This is a particularly critical point in dry environments where lack of water may limit the hydrological requirements of biogeochemical processes removing nutrients in wetlands (Mandi et al., 1998; Lissner et al., 1999b; Comín et al., 2005).

Few experiments have been carried out with constructed wetlands in semiarid zones compared to humid ones (Cerezo et al., 2001; Howell et al., 2005). High temperature and radiation during the growing season are characteristic of semiarid climates, affecting microbial activity and plant productivity, directly or indirectly (e.g., through increased salinity). The relationships between salinity and wetland functioning is a major gap in ecological research. The effects of salinity increase on *Phragmites australis* were studied by Lissner et al. (1999) in experimental conditions, giving relatively good plant growth up to 10 psu (practical salinity units)

Improvement of landscape diversity has also been addressed by constructing wetlands (Knight, 1992b; Comín et al., 2001). Thus, wetlands should be restored and constructed with a suitable size ensuring their integration at landscape scale. Several contributions have been reported providing guidelines for planning at catchment scale. Hammer (1992) recommended choosing an acceptable 2% of the catchment area for wetlands establishment to virtually treat all runoff. Equally, Larson et al. (2000) showed a design that intercepts agricultural drainage with a watershed to wetland ratio of 22:1, meaning that approximately 4.7% of the catchment was covered by wetlands. Finally, it has been suggested that a sufficient improvement of the water quality would be achieved returning between 3 and 5% of the Mississippi catchment area to wetlands (Mitsch et al., 2001). Additionally, modeling incorporates different criteria for restoration suitability which provides a variety of results for wetland restoration potential at watershed scale depending on the selected and weighted variables (White and Fennessy, 2005). Moreover, it has been illustrated that wetland functioning is more effective in upper reaches of a catchment than in lower ones in spite of nutrients concentration of runoff treated is lower in the first (Prato et al., 1995; Woltemade, 2000).

Large areas of the central Ebro river valley (NE Spain) were transformed into irrigated agricultural fields during the 1950s-90s causing soil and water salinization and landscape homogenization (Tedeschi et al., 2001b; Causapé et al., 2004). Nowadays, there is a social demand for increasing landscape diversity, enhancing habitat quality for wildlife and improving river water quality. Accordingly, we established two goals for the present study: estimating the efficiency of newly constructed wetlands on old irrigated fields to remove nitrogen and phosphorus from agricultural wastewater and their potential contribution to increase landscape heterogeneity.

Study area

Monegros is a 270,000 ha region located in the central part of the Ebro River basin, NE Spain (Fig. 1). Climate is semiarid and Mediterranean–continental. Average annual temperature is 14.5ºC; average annual precipitation is 400 mm (Pedrocchi, 1998a) with a high interannual variability (Comín and Williams, 1993). Soils are mainly dominated by a Tertiary structure composed of clays with different salinization levels (conductivity ranges between 1-10 mS cm^{-1}) with inserted sandstone stratus. Locally, this structure is covered by gravels 2-4 meters wide. Most of this land was transformed into irrigated agricultural fields during the 1950s-90s. Soil salinization and abandonment of agriculture is now widespread in many parts of this region.

The original pseudo-steppe vegetation, dominated by Rosemary (*Rosmarinus officinalis*), Thyme (*Thymus* sp.) and halophytes (*Salicornia* sp.) as well as perennial grasses (*Lygeum spartium*, *Brachypodium retusum*) changed into a landscape dominated by new halophytic species (*Tamarix africana*, *Atriplex* sp., *Suaeda vera*) in abandoned dry elevated zones and helophytes (*Phragmites australis*, *Typha latifolia*, *Scirpus holoschoenus*, *S. maritimus*, *Carex divisa*) in the abandoned wetter and lower zones of the valleys. Most of these wetlands are colonized by *Phragmites australis* which is spreading and increasingly dominating the plant community because of its tolerance to changing water level and salinity (Lissner et al., 1999a). *Phragmites* dominated wetlands exist in the lower parts of non-irrigated valleys in the study area; this indicates that this type of wetlands are spontaneously formed at the hydrological discharge zones of the valleys.

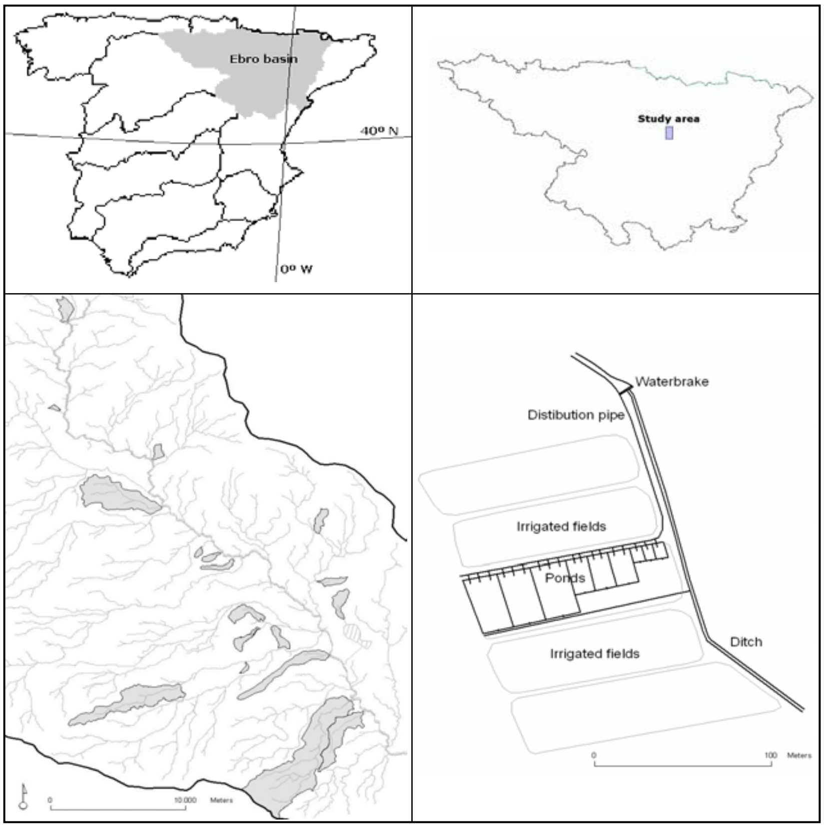

Figure 1. Map of the study area with the main catchments of the Spanish Iberian Peninsula (top left), study area located in the Ebro basin (top right), selected catchments (in grey) included in the Flumen basin and its drainage network (bottom left) and plan of the experimental wetlands (bottom right).

In the study area 19 catchments were selected for the landscape survey. An abandoned field (1.92 ha) that was irrigated in the past, was used to establish experimental wetland plots (Fig. 1). They were banked and flooded with wastewater from the surrounding agricultural irrigated fields to study their effectiveness for water quality improvement.

Material and methods

Constructed wetlands

Nine experimental plots of different size (small: 5×10 m; medium: 10×20 m; large 20×40 m) were established in an abandoned agricultural field and they were operative during the growing seasons 2004–2005 (Fig. 1). Plots were essentially covered by *Phragmites australis* (with scattered stems of *Typha latifolia, Scirpus lacustris* and *Carex divisa*) spontaneously grown since field abandonment four years earlier. They persisted thanks to continuous water contributions from some leaks of a nearby ditch. These leaks were repaired before plots conditioning. Wetlands showed a density of *Phragmites australis* of 165.15±115.57 stems/m^2, height of 152.32±56.34 cm and aboveground biomass of 1.58±1.19 kg/m^2. All plots were kept flooded as long as possible with the same agricultural wastewater that was collected in a ditch with a dike at 100 m distance and conveyed separately to each plot by a distribution pipe (Fig. 1). Wastewater came from soil infiltration of irrigated land in the surrounding fields. Samples of the agricultural wastewater were collected before entering the experimental plots to analyze main water characteristics (Table 1). These characteristics were determined in the same way as those colleted at inflow and outflow of the plot. Inflowing wastewater was regulated in each plot by 1 to 3 valves in order to avoid flow concentration (Reed et al., 1995). Water depth was fixed at 10 cm allowing a water-residence time between 1 and 4 days. Nevertheless, the climatic conditions and the lack of wastewater made often difficult to maintain a constant water level and water turnover. Water flow was estimated after the time needed to fill a container of known volume. Since the hydrologic regime depended on watering periods, it was difficult to completely provide a constant flow to the experimental plots.

Samples of inflowing and outflowing water were collected in all the plots three times in 2004 and four times in 2005 during the growing season. In situ field measurements of temperature, pH, dissolved oxygen (DO), and electrical conductivity (EC) were performed with calibrated electronic apparati. Water samples were filtered (0.8 µm mesh size pre-combusted filters) the same day of sampling, and total dissolved solids (TDS), total suspended solids (TSS), turbidity, alkalinity, Cl^-, Ca^{2+}, Mg^{2+}, Na^+, and K^+ were measured using standard methods in the laboratory (APHA, 1998) Micronutrients were analyzed from frozen (at -30ºC) sample aliquots one month after sampling using standard methods (APHA, 1998). Soil salinity was estimated as EC of a saturated paste made of 1:5 proportion of soil and distilled water. ANOVA and Kruskal-Wallis tests were used to test for statistical differences between inflow and outflow water characteristics, and for differences of nutrient retention between plots of different size.

	Units	Max	Min
pH		8.21	7.51
EC	µS	1165.00	498.00
DO	mg/l	11.89	5.58
TSS	µg/l	3.20	1.07
TDS	mg/l	772.00	544.00
Chloride	mg/l	176.00	80.80
Sulphates	mg/l	123.31	47.20
Ca^{2+}	mg/l	140.90	73.20
Mg^{2+}	mg/l	27.00	17.90
Na^+	mg/l	226.25	52.03
K^+	mg/l	2.04	0.00
Turbidity	mg Pt/l	54.00	3.00
Alkalinity	mg CaCO3/l	362.40	259.10
NO3 - N	mg/l	20.65	5.80
NO2 - N	µg/l	61.71	0.00
Ammonium	µg/l	190.21	16.99
Org N	mg/l	16.12	0.00
TN	mg/l	22.82	16.11
SRP - P	µg/l	98.87	3.17
Org P	µg/l	177.89	34.91
TP	µg/l	276.76	38.93

Table 1. Characteristics of the wastewater running off the surrounding irrigated fields at the wetlands inflow. Maximum and minimum values are given only because of the high variability of the data along the year.

Existing wetlands

All wetlands larger than one hectare located in catchments of irrigated farms of the Flumen River and included in Monegros area were selected. In addition to them, six more catchments with wetlands smaller than one hectare and five more with no representative wetland surfaces were also selected for study, i.e. 19 (in all). The criteria followed to select these additional catchments were the topographical, geological and geographical similarity to those with large wetlands (Fig. 1 and 2). All 19 catchments were digitalized in ArcGIS 8.3 (ESRI, Inc.) with aerial photographs 0.5 m resolution taken in 2003. Land cover types in the photographs were checked in situ by direct observation and their areas estimated using ArcGIS 8.3. The considered land cover types were arboreal vegetation, dry scrublands-grasslands, erosion deserts, urbanized areas, dry farmlands, irrigated farmlands, abandoned farms, arboreal cultivations, wetlands, channels, livestock farms and irrigation ponds Then, landscape spectral diversity was estimated using homogeneous segments of 6 ha (300×200 m) distributed along the major geographical axis of the catchment. We used the Shannon diversity index (H = - $\sum p_i \cdot \log_{10} p_i$, where p_i is the area of landscape type i / total segment area) to estimate landscape heterogeneity in each catchment. Spectral landscape diversity was calculated by accumulating areas along each transect. Obtained diversity spectra of catchments were grouped by sizes in the same way as done in Figure 2. This calculation gives us information about the spatial heterogeneity of the landscape diversity (Margalef, 1974; Turner et al., 1989; Comín et al., 2001).

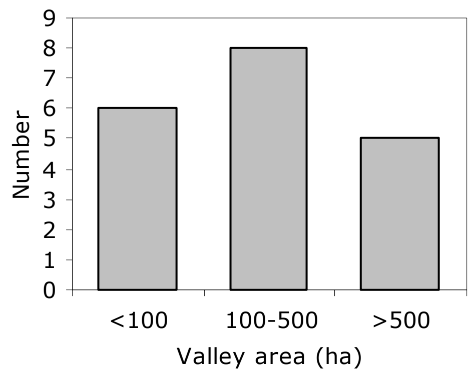

Figure 2. Number of selected valleys studied for landscape characteristics sorted in three size groups.

Results

Water quality

The water entering the experimental wetlands was fresh with relatively high dissolved salt content and alkalinity. It was typical agricultural wastewater, with nitrate and phosphate concentrations over those advised by European Directives (EC, 2000).

A clear difference between nitrogen concentration at wetland inflow and outflow was observed (Fig. 3). Total nitrogen (TN) decreased from 16–19 mg/l in the inflowing water to 12–16 mg/l in the outflowing water. Most of TN (78–90%) was dissolved inorganic nitrogen in form of nitrate (NO_3-N) that decreased from 70–83 mg/l to 46–63 mg/l. This facilitated nitrogen removal by biogeochemical processes that reached 24–43% of the total inflowing nitrogen in the second year (with a peak of 98% in June 2005). No significant differences of nitrogen retention between plot sizes were observed after ANOVA analysis (P=0.05). However, a trend relating

increasing nitrogen retention, both NO_3-N and TN, to the size of the plot is observed for the full operating period (Fig. 4). The second year of wetland functioning showed a higher retention for TN and NO_3–N than the first with similar inflow concentrations (Fig. 3 and 5). In summer with a TN inflow concentration of 20 mg/l, retention increased from 19% the first year to

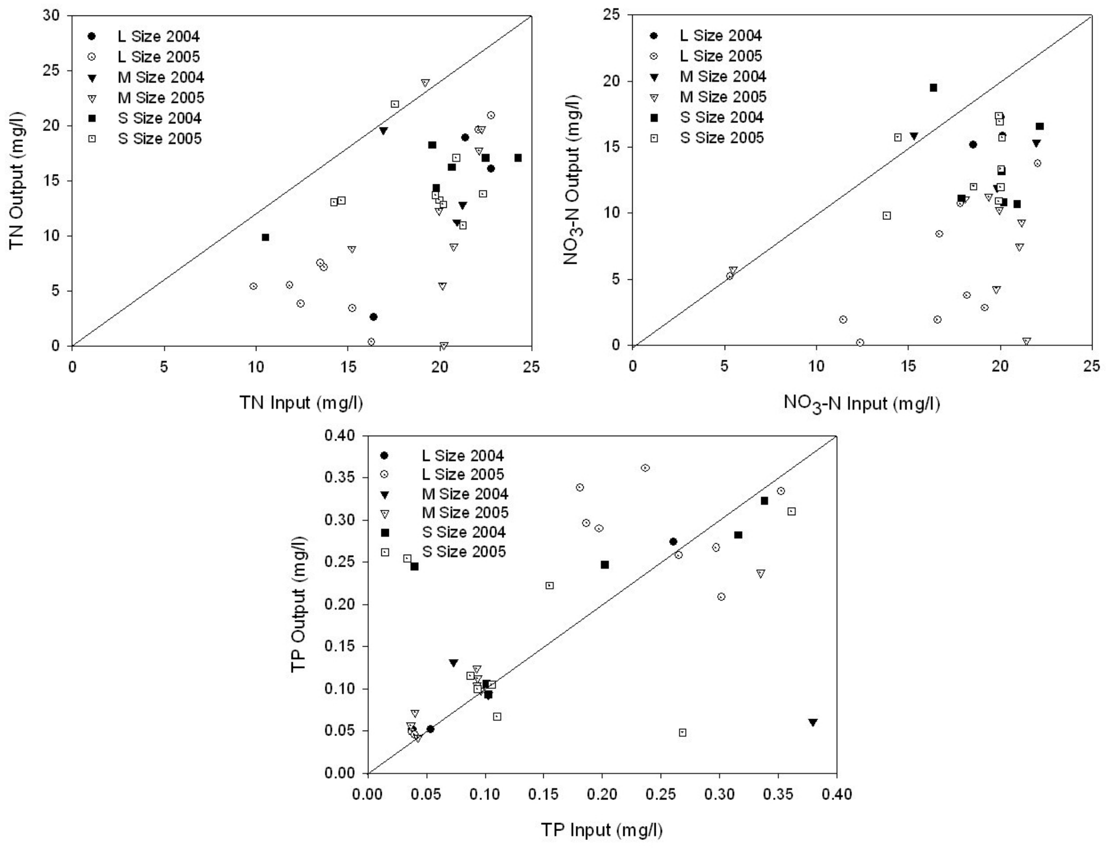

Figure 3. Input/output relation for samplings taken the same day for total nitrogen (TN; top) and dissolved inorganic form of nitrate (NO3–N; medium)for years 2004 and 2005 and by wetland area (S–50 m2, M–200 m2 and L–800 m2). Input/output relation for total phosphorus (TP; bottom).

52% the second year. In autumn, with a similar TN inflow concentration, removal changed from 12% the first year to 41% the second year. Figure 5 shows also a severe drop in winter of NO_3–N in regards to TN that remained at similar concentration values during all the year. This means an increase of the proportion of organic N, probably due to the reduction of fertilizer contributions and the increase of decomposition of crop remains.

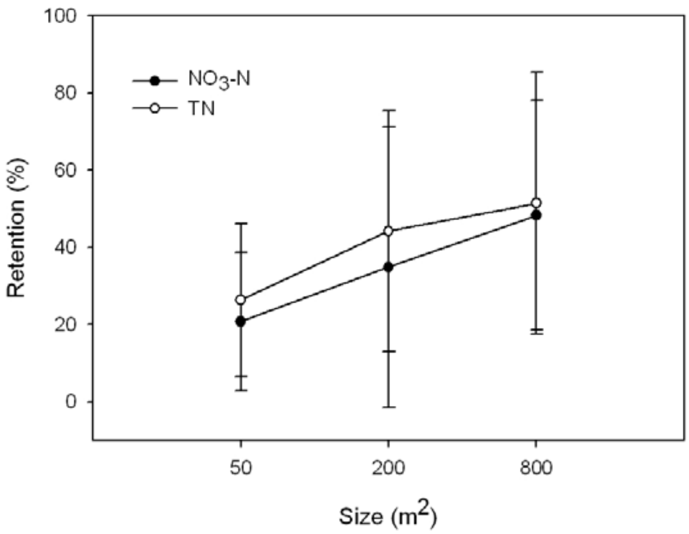

Figure 4. Retention percentage of TN and NO_3–N in relation to wetland area.

Phosphorus removal efficiency was very low compared to nitrogen in these wetlands. Export or no net change was observed during half of the observations in all plots, with no significant differences between plot sizes (Fig. 3). After two years of wetland functioning nutrients removal was not influenced by water-residence time which was entirely dependent on watering regime of upper farms. Additionally, no relation could be established between water-residence time and plot size.

No significant changes in pH, EC, DO, TDS, TSS, turbidity, alkalinity, Cl^-, Ca^{2+}, Mg^{2+}, and K^+ were observed between inflow and outflow in the plots. There were no significant differences between inflow and outflow concentration of sodium either (12.92±20.52 %).

Wetlands features and landscape diversity

All the wetlands existing in our study area used for the landscape approach were located in catchments mostly occupied by irrigated agricultural fields (67-70% of their total areas). These wetlands had different forms (elongated, branched, lobed, amoeboid, and irregular) and were located in different positions in the catchments (valley bottoms, streams, and hillsides). Most of these wetlands were smaller than 500 ha and just one of them was larger than 1,000 ha (Fig. 2). The area of these wetlands is 1.5–4% of their respective catchment areas. The overall landscape diversity was low (maximum 0.72). Higher values were observed in larger valleys, although little differences were found in the heterogeneity between sizes of catchments (Fig. 6). Large catchments showed cases with low diversity values at the head that increased along the transect (e.g. V08 in Fig. 6), and other cases that were high at the head and remained constant along the transect (e.g. V19 in Fig. 6). Medium catchments had also cases that showed low diversities at the head and after increased more or less quickly (e.g. V12 and V16 in Fig. 6). The rest of them had low values since the beginning, remaining constant and even decreasing advancing towards the mouth (e.g. V03 and V06 in Fig. 6). In small catchments diversity had mostly diagonal evolution, providing certain heterogeneity.

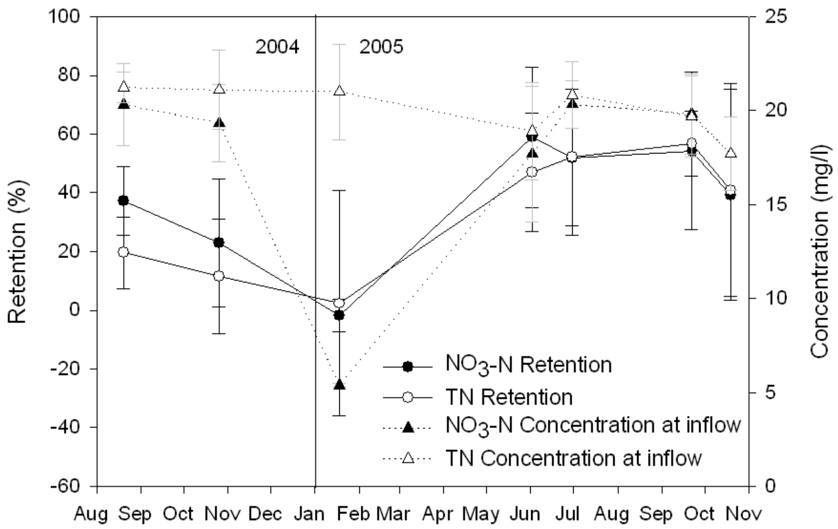

Figure 5. Seasonal changes of TN and NO3–N removal percentage and TN and NO3-N concentration at inflow during the study period.

We observed four different types of diversity spectrum: a) a rectangular type, which was characterized by reaching the maximum diversity at the proximal zones of the transect (e.g., V19 in Fig. 6), b) inverse rectangular type, characterized by very low diversity for most of the transect length but increasing up to the maximum diversity value after integration of the last parts of the transect (e.g., V12 and V08 in Fig. 6), c) a horizontal type, characterized by an almost constant diversity for all the diversity spectrum (e.g., V06 in Fig. 6), d) a diagonal type, indicated a heterogeneous landscape increasing its diversity along the major axis of the valley (e.g., V17 in Fig. 6). The first diversity spectrum type was representative of catchments which still preserved untransformed zones at the head of the

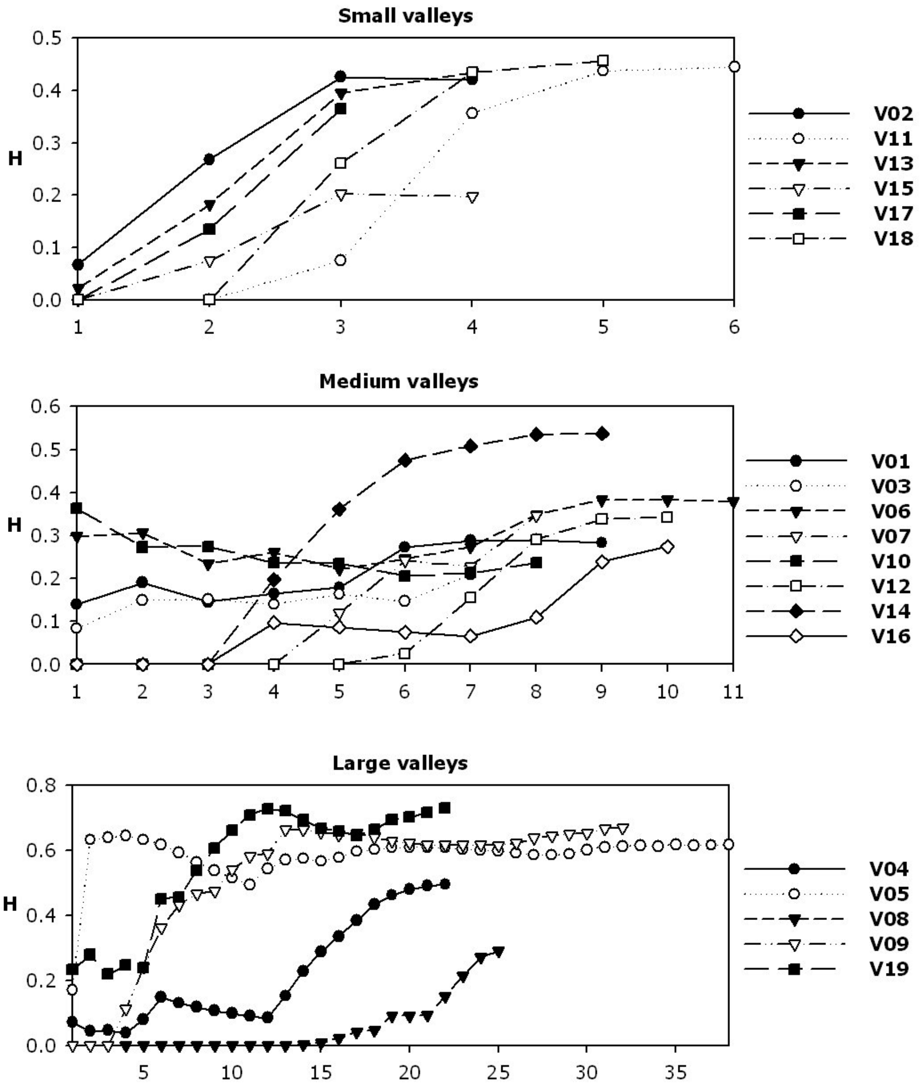

Figure 6. Spectral diversity of transects of the selected catchments. H refers to landscape heterogeneity of a catchment (Shannon index). Horizontal axis indicates the number of each segment and represents cumulative values of the transect area studied from the head to the mouth of the catchment.

catchment (first areas of the transect). The second spectrum type was typical of catchments extensively transformed into irrigated fields, but still preserving wetland zones at lowest part of the catchment which coincided with the last parts of the transect. The third diversity spectrum type corresponded to catchments with extensive irrigation use but also transformed physically to flat the terrain and facilitate this land use. The fourth type of spectrum was characteristic of small catchments which still preserve some landscape heterogeneity.

Discussion

This study is an example of how constructed wetlands in a dry environment mitigate the problems related to non-point pollution from irrigation of agricultural fields. Our results encourage this approach to reduce the nutrient enrichment of water and increase landscape diversity.

Our experimental wetlands showed a remarkable N removal efficiency (24–43% of the inflowing N concentration), in spite of the fact that they were at the initial stages of their development (two years). For phosphorus, the experimental wetlands ranged from exportation to removal of 80% of the phosphorus input. These removal efficiencies are within the ranges of other case studies. Mitsch and Gosselink (2000) reviewed the results for a number of wetlands and reported a range of 40-95% of N retention and from exportation to 99% retention of phosphorus. Comin et al. (2001) reported a range of 50-98% of N retention in *Phragmites* dominated wetlands after four years of operation in similar climatic conditions. In a boreal zone, Koskiaho et al. (2003) measured 0–36% N retention with lower residence times in the wetlands (6-24 h) than in our study, and no significant results for phosphorus.

The hydrologic loading of these wetlands fluctuated and was entirely dependent on climatic conditions which regulated the water available for irrigation and, consequently, the amount of agricultural wastewater. Therefore, water residence time was very variable in our experimental wetlands (usually between one and four days). In addition, the exceptional severe drought of 2005 maintained the wetlands dry (with no surface water) for several periods during the growing season and very likely reducing removal efficiencies. These flow characteristics may have determined the lack of relationship between nutrient removal and water residence time or pond size. Similar situations have been studied submitting wetlands to drying and rewetting periods. An increase of extractable phosphorus release has been reported after rewetting the wetland soils (Chepkwony et al., 2001; Venterink et al., 2002).

The effect of plots size showed a trend, although not significantly. Larger wetlands (800 m^2) removed more N than medium (200 m^2) and small ones (50 m^2). Some differences between the first and second years of wetlands operation were observed. An increase of TN retention from 10-20% the first year to 40-50% the second year has been detected for similar inflow concentrations of TN. This means that these wetlands are progressing in their nutrient removal function towards and advanced state with high nutrient removal efficiency as it is usual in similar wetlands (Mitsch and Gosselink, 2000a; Comín et al., 2001).

Landscape diversity slightly increased with catchment area although the maximum landscape diversity was very low (0.72) compared to other landscapes (Eiden et al., 2000). No relationship was observed between catchment area and type of landscape diversity spectrum. Large catchments only show rectangular and inverse rectangular diversity spectra. Medium size catchments show horizontal and inverse rectangular spectra. Landscape diversity spectra of small catchments are diagonal. This lack of relationship may be attributed to the extensive land use cover change in favor of irrigated agriculture which has been a common fact all around the study area and has eliminated scale dependent landscape patterns. The result of this land use cover change has been an intensive landscape homogenization.

The expected diversity spectrum in a heterogeneous landscape is a diagonal one which would correspond to catchments with a gradient of landscape types along the major geographic axis in our study areas. It was only observed in small catchments which does not allow to incorporate large irrigated areas in the spectrum while maintaining a mosaic of land use cover types along the transect. This gives spatial heterogeneity to the overall catchment. Compared to this relatively heterogeneous catchment type, those with other type of diversity spectrum have a relatively homogeneous landscape. This is a consequence of extensive land use cover changes performed at the bottom of the valley (rectangular spectrum), at the head of the valley (inverse rectangular spectrum), or all along the valley (horizontal spectrum).

In general, wetlands are located in the bottom of the valleys as a consequence of surface run-off and/or groundwater discharges. Thus, restoration or construction of wetlands will strengthen an increase of diversity at lower parts of the catchments provided they were landscape integrated at suitable sizes.

The experimental constructed wetlands covered 1.4% of its watershed where and they were able to remove up to 50% of the total nitrogen (which is the most critical pollutant in the zone) of the agricultural wastewater during the second year of functioning with an alternately (dry/flood) retention time of 1-4 days. It is expected that this efficiency will improve after a few years of wetland functioning. In any case, increasing the wetland area up to twice that of our experiments (approximately 3% of the total catchment area) will ensure a high removal efficiency of nitrogen by wetlands. This area (3%) is of the same order to those proposed by others to play a high efficient role to treat non-point source pollution in agricultural watersheds (Hammer, 1992; Larson et al., 2000; Mitsch et al., 2001).

In addition to remove nutrients, restored wetlands can contribute the added value of increasing landscape diversity if they are distributed in different zones and not only at the bottom of the catchment, thus decreasing the homogeneity of areas extensively transformed by agricultural activities.

Acknowledgements

Thanks are given to Sindicato de Riegos de Orillena, particularly to its president J. Pujol for his on-going assistance and information. Field and laboratory work was assisted by A. Supervía, J. Cervantes and G. Martínez. J. M. Rey Benayas (Alcalá de Henares University) and two anonymous reviewers are acknowledged for their useful comments. This work was supported by CYCIT (REN2003-03040) and DGA–Research Group on Ecological Restoration.

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Chapter 3

Effects of wetlands construction on water quality in a semi-arid catchment degraded by intensive agricultural use*

Abstract

Many factors can influence the improvement of water quality in surface-flow constructed wetlands (SFW). To understand how water quality changed after passing through SFW, 11 plots of various sizes (50, 200, 800m^2 and 5000 m^2) were established within constructed wetlands on agricultural soils in the Ebro basin (NE Spain) that were affected by salinization. A set (n=16) of water-quality parameters (e.g., nutrients, salts, sediments, and alkalinity) were sampled at the inflow and outflow of the wetlands during the first four years after the wetlands were constructed. NO_3-N retention rates were as high as 99% in the large (5000 m^2) wetlands. Even after four years, TP and salts were released from the wetlands. Over the same period, in small wetlands (50, 200, and 800 m^2), NO_3-N retention increased from 30% to almost 60%, and Na^+ release declined from 35% to 9%. In addition, the large wetlands exported C. At the current rate of NO_3-N retention (50-80 g·m^{-2}), 3.5-6.5% of the catchment should be converted into wetlands that are closer to 5000 m^2 than to 800 m^2 in size and that have water-residence times of about 4 d.

Resumen

Muchos factores pueden influir en la mejora de la calidad del agua en humedales construidos de flujo superficial. Para entender como cambia el agua después de atravesar un humedal de flujo superficial, se establecieron 11 humedales de este tipo de varios tamaños (50, 200, 800m^2 y 5000 m^2) sobre suelos agrícolas en la cuenca del Ebro (NE España) afectados por procesos de salinización. Una serie (n=16) de parámetros de la calidad del agua (nutrientes, sales, sedimentos y alcalinidad) fueron muestreados a la entrada y salida de los humedales durante los primeros cuatro años tras la construcción de los humedales. La retención de NO_3-N alcanzó el 99% en los humedales más grandes (5.000 m^2). Incluso después de cuatro años de funcionamiento, se liberaban fósforo y sales desde los humedales. En el mismo periodo, en los humedales pequeños (50, 200, 800m^2), la retención de NO_3-N se incrementó de un 30% a un 60% y la liberación de sales

* Moreno, D., Comín, F.A. and Pedrocchi, C. 2008. *Ecological Engeneering*. In press.

descendió de un 35% a un 9%. Además, los humedales grandes exportaron carbono. A la proporción actual de retención de NO_3-N (50-90 $g \cdot m^{-2}$), entre el 3.5 y el 6.5% de la cuenca debería ser convertido en humedales con tamaños más cercanos a 5.000 m^2 que a 800 m^2 y con periodos de retención de agua de unos cuatro días.

Introduction

In agricultural landscapes, the control of non-point pollution is a major concern and a significant challenge (Mitsch and Gosselink, 2000a). The most commonly advocated means of reducing non-point pollution is to restrict the application of fertilizers upstream to those that are absolutely necessary (Causapé et al., 2004; Verhoeven et al., 2006) or to construct ecosystems that remove pollutants from the agricultural runoff (Hammer, 1992; Raisen and Mitchell, 1995; Zedler, 2000). Surface-flow constructed wetlands (SFW) are an efficient and economical tool to control non-point pollution in large agricultural areas (Mitsch et al., 2001; Kovacic et al., 2006).

The use of SFW to control non-point pollution in agricultural catchments has been studied across a wide range of climatic conditions, including boreal (Braskerud, 2002b; Koskiaho and Puustinen, 2005), temperate (Woltemade, 2000; Trepel and Palmeri, 2002; Jiang et al., 2007; Scholz et al., 2007), tropical (de Ceballos et al., 2001; Chimney and Goforth, 2006), and semi-arid zones (Comín et al., 2001; Howell et al., 2005). Under those conditions, many environmental variables (e.g., temperature, hydraulic load, pollutant concentrations, soil properties, plant cover, microbial activity) and features of wetland design (e.g., water-residence time, depth, shape, plant species composition) can influence retention rates, permitting sometimes the release of nutrients (in particular, P). The construction of wetlands on agricultural soils can have a significant effect on water quality. Of particular importance is the dynamic of the Ca-P system that allows the release of P from the soil into the water, or the precipitation of Ca and P, which depends on factors such as pH and the concentration of Ca (Van den Berg and Loch, 2000; Shenker et al., 2005). After inundation, agricultural soils can release nutrients (N and P) into the water outflow, which progressively decreases their concentrations (Bruland et al., 2003). In other circumstances, recently created wetlands can quickly develop the capacity to remove nitrogen from wastewater (Romero et al., 1999; Fink and Mitsch, 2004; Borin and Tocchetto, 2007; Moreno et al., 2007).

In the field of wetlands restoration, the effect of soil salinity on water quality is not well studied. The effects of the release of salts caused by irrigation are well known because of the important drawbacks caused on the agricultural runoff (Tedeschi et al., 2001b; Causapé et al., 2004). In addition, salinity can affect plant growth (Lissner et al., 1999a) and nitrogen mineralization (Irshad et al., 2005). Preliminary research in the study area indicated that surface flow wetlands rehabilitated on old agricultural fields were able to remove significant amounts of nitrogen but not so much phophorus from agricultural wastewater (Romero et al., 1999; Comín et al., 2001), and no export of major salts from salinized soils took place (Moreno et al. 2007).

The general aim of this study was to estimate the changes of the agricultural wastewater parameters passing through SFW constructed upon old irrigated agricultural fields. For this retention/release rates of nutrients (N and P), salts and sediments were estimated for 16 variables recorded at inflow and outflow channels. Interactions between these variables let explain patterns of retention/release. Also requirements for efficient establishment of SFW to remove nutrients and salts from agricultural wastewater at catchment scale are discussed, which is a major issue for wetland creation and restoration (Mitsch and Gosselink, 2000b; Zedler, 2000).

Study area

Monegros is a 270,000-ha region in the centre of the Ebro River Basin, NE Spain (Fig. 1). The climate is semiarid and Mediterranean–continental. Average annual temperature is 14.5ºC and average annual precipitation is 400 mm (Moreno et al., 2007), but interannual variability is high (Comín and Williams, 1993). The soils mainly are dominated by a Tertiary structure that consist of clays that have variable levels of salinity (conductivity=1-10 mS cm^{-1}) with inserted sandstone strati. Locally, the structure is covered by gravels that are 2-4 m wide. Most of the land was converted into irrigated agricultural fields during the second half of the 20th C. Soil salinization and abandonment of agriculture is now widespread in many parts of the region. The original pseudo-steppe vegetation, which is dominated by Rosemary (*Rosmarinus officinalis*), Thyme (*Thymus* sp.), and halophytes (*Salicornia* sp.), as well as perennial grasses (*Lygeum spartium*, *Brachypodium retusum*), has changed into a landscape dominated by halophytic species (*Tamarix africana*, *Atriplex* sp., *Suaeda vera*) in abandoned, dry elevated zones, and by helophytes (*Phragmites australis*, *Typha latifolia*, *Scirpus holoschoenus*, *S. maritimus*, *Carex divisa*) in the abandoned wetter and lower zones of the valleys. Most of the wetlands have been colonized by *P. australis*, which is spreading and increasingly dominating the plant community because it tolerates fluctuations in water levels and salinity (Lissner et al., 1999a). In the study area, the *Phragmites*-dominated wetlands are in valley-bottoms of non-irrigated valleys, which suggest that these wetlands develop naturally within the hydrological discharge zones of the valleys.

Constructed wetlands

In 2004, 11 experimental plots of various sizes [5×10 m^2 (S), 10×20 m^2 (M), 20×40 m^2 (L), and ~ 5,000 m^2 (H); Fig. 1] were established in abandoned agricultural fields and they were operative during the growing seasons between 2004 and 2007. The fields were within a 22.35-ha catchment that consisted entirely of irrigated farmland. The plots were in fields that were virtually covered by *P. australis* (with scattered stands of *Typha latifolia, Scirpus lacustris*, and *Carex divisa*), which had grown naturally after the fields were abandoned four years earlier. Wetland vegetation persisted because water leaked continuously from a ditch nearby. Those leaking points were closed before wetland conditioning. Between 2004 and 2006, the mean (± SD) density, height, and biomass of *P. australis* increased from 213.5 (64.4) stems/m^2, 88.3 (24.0) cm, and 0.84 (0.39) kg/m^2, to 599.8 (205.6) stems/m^2, 193.2 (50.5) cm, and 3.00 (1.02) kg/m^2, respectively. All of the plots were flooded for as long as possible with agricultural wastewater collected in a ditch that had a dike 100 m distant and was conveyed to each plot separately by way of by distribution pipes (Fig. 1). The wastewater came from soil infiltration of the surrounding irrigated fields. Evapotranspiration (ET) was estimated using a pan where original wetland soil (with *Phargmites* stems and rhizomes) where was planted and flooded until a marked level. After prescribed period, the new water level was measured and the pan was refilled until the marked level. That process was repeated three times per year. The estimated ET for the growing season (April to September) was 6.50±3.32 mm·month^{-1}.

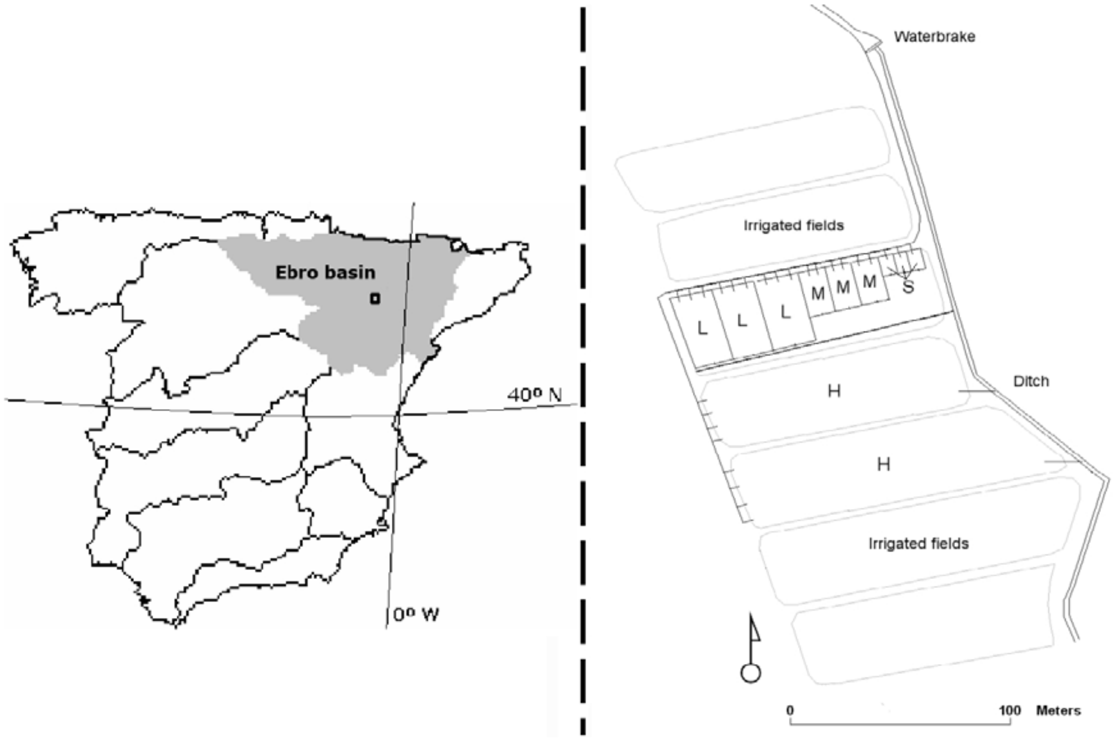

Figure 1. Map of the study area with the main catchments of the Spanish Iberian Peninsula (right) and design of the experimental wetlands (left)

To avoid flow concentration, in each plot, one to three valves were used to regulate inflowing wastewater (Reed et al., 1995). Water depth was fixed at ~10 cm. The climatic conditions and the lack of wastewater often made it difficult to maintain a constant inflow, and water turnover and water-residence times (WRT) were highly variable (Table 1). In 2005, a severe drought kept some of the ponds dry for long periods (up to two months). In those circumstances, the largest ponds were the first to have their water supply cut off. In every year of the study, less severe and punctual water restrictions occurred, which precluded a constant inflow and the permanent flooding of all of the wetlands. On every sampling day, WRT was estimated using the water inflow, as the time it took to fill a container of known volume, and the volume of water stored in the plot (water depth x plot surface), measured every sampling day in a calibrated pole.

Size (m^2)	Year	WRT
50	2004	0.70±0.16
	2005	2.24±0.40
	2006	3.48±0.88
	2007	2.12±0.40
200	2004	2.38±0.27
	2005	4.18±1.34
	2006	4.43±1.94
	2007	2.11±0.28
800	2004	10.59±3.21
	2005	5.64±0.86
	2006	3.06±0.74
	2007	2.31±0.50
5,000	2004	31.16±9.62
	2006	8.69±0.86
	2007	8.84±3.17
General	—	5.02±1.64

Table 1. Water-residence time (WRT; mean±SE) per year and size. General represents WRT mean±SE for the complete study period and for all the wetland sizes.

Materials and methods

Water sampling

To quantify the characteristics of the agricultural wastewater, samples were collected before the water entered the experimental plots (Table 2). In the growing season (April to October), three (2004), four (2005, 2006), or five (2006) samples of inflow and outflow water were collected from each of the wetlands plots. Drought caused the sampling effort to vary among years. Calibrated electronic devices were used to record in situ measurements of water temperature, pH, dissolved oxygen (DO), and electrical conductivity (EC). Water samples were filtered (0.8 µm mesh pre-combusted filters) on the day they were collected, and total dissolved solids (TDS), total suspended solids (TSS), turbidity, alkalinity, Cl^-, Ca^{2+}, Mg^{2+}, Na^+, and K^+ concentrations were measured using standard methods in the laboratory (APHA, 1998). One month after the samples were collected, standard procedures (APHA, 1998) were used to quantify micronutrients in frozen (T = -30 ºC) aliquots. Soil salinity was estimated based on the EC of a saturated paste derived from a 1:5 ratio of soil and distilled water. In addition, to estimate the total contribution of pollutants from the catchment to the hydric network the catchment, between 2004 and 2006, one sample per year was collected just before the water entrance to wetlands. Every time the water was sampled, the flow of the catchment was measured using the principle of the conservation of salt (Comín et al., 2001).

	Units	Mean	Max	Min
pH		7.84	8.21	7.48
EC	dS/cm	1039.00	1172.00	519.00
DO	mg/l	7.74	11.47	5.23
TSS	µg/l	1.68	10.60	0.27
TDS	mg/l	671.07	900.00	388.00
Chloride	mg/l	107.38	213.99	38.60
Sulphates	mg/l	111.70	171.22	44.54
Ca^{2+}	mg/l	106.88	149.30	55.62
Mg^{2+}	mg/l	20.98	29.10	5.00
Na^{+}	mg/l	71.86	219.92	31.67
K^{+}	mg/l	1.36	4.44	0.00
Turbidity	mg Pt/l	32.69	330.00	0.00
Alkalinity	mg CaCO3/l	247.58	508.80	143.70
NO_3–N	mg/l	19.90	33.70	0.02
SRP–P	µg/l	2.32	21.36	0.00
TP	µg/l	5.42	26.50	0.00

Table 2. Characteristics of the wastewater running off the surrounding irrigated fields at the wetlands inflow. Maximum and minimum values are given because of the high variability of the data along the year.

Spearman's Rank Correlation coefficients were used to evaluate the relationships among water-quality parameters and WRT. Variables that were strongly correlated ($p<0.01$) with two or more variables were excluded from further analyses. The distributions of the data were skewed; therefore, non-parametric tests were used. Kruskal-Wallis Tests were used to test for statistical differences in the characteristics of inflow and outflow waters, and in nutrient retention among plots of different sizes. To determine which plot sizes accumulated the most variance, orthogonal contrasts were performed. For all of the tests, the p-value for statistical significance was set at $p<0.05$. All of the statistical analyses were performed in SPSS for Windows 13.0 (SPSS Inc., 2004).

Results

In the study area at Monegros, in the Ebro River Basin, Spain, the concentrations of nitrate and phosphate in the wastewater at the inflow were higher than those recommended by the European Commission (NO_3-N: 10 mg·l^{-1})(European-Communities, 2000). In most respects, the waters at the inflow and outflow differed significantly (Table 3; Fig. 2). The retention rates of some of the water parameters differed significantly between plot sizes S-M-L and plot size H. Collinearity was significantly (p<0.01) high (1) between Cl^-, SRP-P, alkalinity, and DO, (2) between TDS, alkalinity, and EC, and (3) between SRP-P, TP, and NO_3-N,. Therefore, Cl^-, TDS, and SRP-P were excluded from the analysis. In the S, M, and L wetlands, the retention rates of NO_3-N by volume varied between 30.39 ±7.10% (mean ± SE) in 2004 to 41.23 ±5.52% in 2007. In 2007, concentration of NO_3-N passed from 18-19 $mg \cdot l^{-1}$ at the inflow to 8-10 $mg \cdot l^{-1}$ at the outflow (Fig. 2). In 2007, N-retention as measured by mass was 51-88 $gN \cdot m^{-2}$. In the largest wetlands, retention rates of NO_3-N by volume reached 99.03±0.36% throughout 2007. In that period, the water leaving the H wetlands had ~1 $mg \cdot l^{-1}$ (Fig. 3). In 2007, the amount of N removed from the H wetlands was 45-83 $gN \cdot m^2$. TP was significantly released from wetlands throughout the study and was highly variable (Fig. 2; Table 3). The amount of TP at the outflow (6-8 $\mu g \cdot l^{-1}$) was 68.32±37.33% higher that it was at the inflow (4-6 $\mu g \cdot l^{-1}$), which reflects an annual release (by mass) of between 0.43 and 2.99 $gP \cdot m^{-2}$. TP release tended to be higher in the larger wetlands plots (Fig. 3), but the relationship was not statistically significant. In the area of the wetlands, the average N-release by the catchment was 731.34 kg.

	All	2004	2005	2006	2007	Outflow/inflow
pH	***	ns	***	***	***	↑
EC	ns	**	ns	**	*	↑
DO	***	**	***	***	***	↓
Alkalinity	**	**	ns	***	***	↑
Sulphate	ns	<0.1	ns	ns	ns	-
Ca^{2+}	ns	ns	**	ns	ns	-
Mg^{2+}	**	ns	**	<0.1	ns	↑
Na^+	***	ns	**	<0.1	*	↑
K^+	ns	ns	***	<0.1	ns	-
NO_3–N	***	***	***	***	***	↓
TP	*	ns	ns	ns	*	↑

Table 3. Results of the Kruskal-Wallis analysis of the wastewater parameters for the difference between inflow and outflow of the wetlands (*p<0.05; **p<0.001; ***p<0.0001; ns=no significant). Outflow/inflow represents the increase (↑) or decrease (↓) of parameters at outflow regarding to inflow. In this column only parameters significantly different for all years together or for more than one year separately are represented.

The concentrations of salinity-related parameters were higher at the outflow than they were at the inflow (Fig. 2). The values of those parameters in the H wetlands, where flooding was much more irregular, were highly variable; therefore, the data from those wetlands were analyzed separately. In the S-M-L-wetlands, the release of Na^+ by volume decreased progressively from 34.15 ±11.41% in 2004 to 8.82 ±3.12% in 2007 based on an inflow

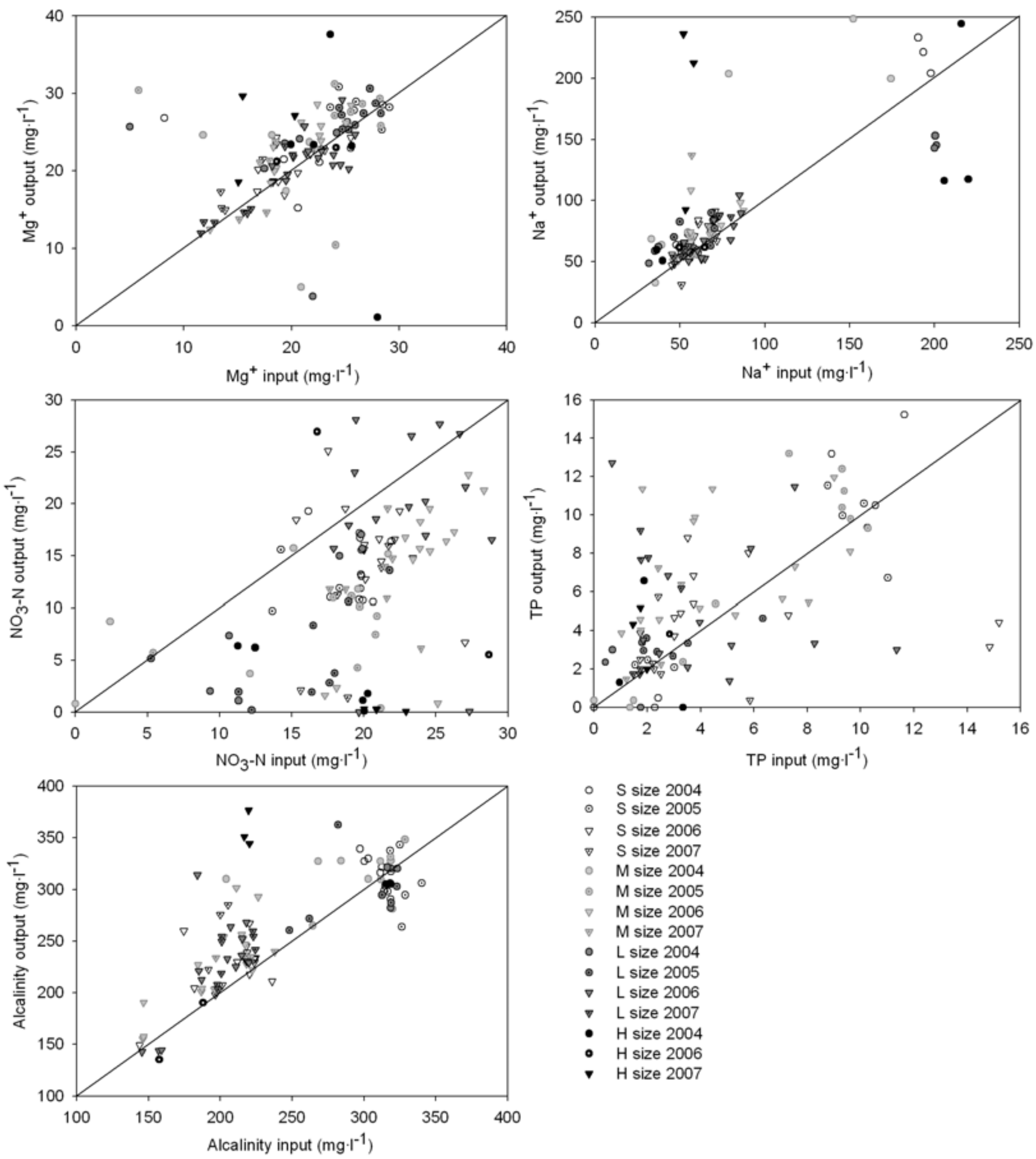

Figure 2. Inflow/outflow concentrations of water quality parameters detected as significant ($p<0.05$) by the Kruskal-Wallis analysis. Samples at inflow and outflow were taken the same day (S–50 m2, M–200 m2, L–800 m2 and H-5,000 m2)

concentration of 59-63 mg·l^{-1} (Fig. 4). By mass, the release of Na^+decreased from 111-515 g·m^{-2} in 2004 to 67-244 g·m^{-2} in 2007. The Na^+ release tended to be higher in the larger wetlands (Fig. 3), but therelationship was not statistically significant. The release of Mg^{2+} decreased progressively from 36.47 ±27.22% in 2004 to 13.90 ±7.07% in 2007 based on an inflow concentration of 20-22 mg·l^{-1}. In 2007, the release of Cl^- was significant (13.55±5.22%; Table 3), but it did not decline progressively throughout the study. In 2005, a significant release of all of the cations took place, including Ca^{2+} and K^+, which were not significantly released in any other year of the study (Table 3). In the S-M-L-wetlands, EC was significantly higher at the outflow than it was at the inflow in most of the years, but the maximum annual average was only ~2% (in 2007). In H-wetlands, only those variables that differed significantly from those in the S-M-L-wetlands were included in the analysis. In the H-wetlands, EC was significantly (40.20±14.39%) higher at the outflow than it was at the inflow (1022-1057 $dS \cdot cm^{-1}$). After the water passed through the H wetlands, the concentration of Ca^{2+} decreased 16.06±7.01% from the concentration in the inflow wastewater (103-111 $mg \cdot l^{-1}$).

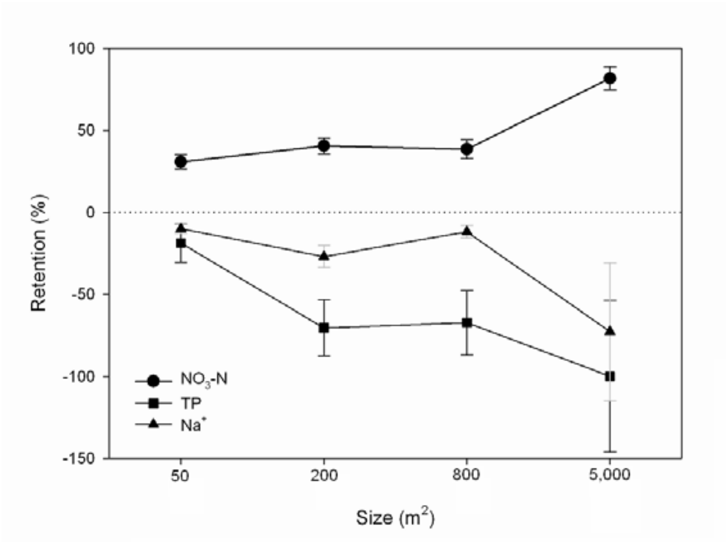

Figure 3. Evolution of the retention with increasing wetland sizes of the main water contaminants. The parameters represented were significant ($p<0.05$) in the Kruskal-Wallis analysis for the factor size. Symbols represent mean±SE.

Alkalinity and pH were significantly higher at the outflow than they were at the inflow (Table 3; Figure 2). The values of those parameters in the H-wetlands, where flooding was much more irregular, were highly variable; therefore, the data from those wetlands were analyzed separately. Alkalinity and pH at the outflow were only 10.67±1.97% and 5.40±0.65% higher, respectively, than they were at the inflow (ph=7.84; $CaCO_3$ concentration=236-259 $mg \cdot l^{-1}$). In 2007, the amount of CO_3^{2-} released by the H wetlands was almost four times higher at the outflow than it was at the inflow.

NO_3-N retention and WRT were significantly positively correlated. Throughout the high retention (~99%) episode in the H wetlands in 2007, the average WRT was ~8 d (range=4-14 d). In addition, the Ca^{2+} retention rate was positively correlated WRT, particularly in the H wetlands, which had long WRT (11.16±2.19 d). TP and sulphate release were positively correlated with WRT.

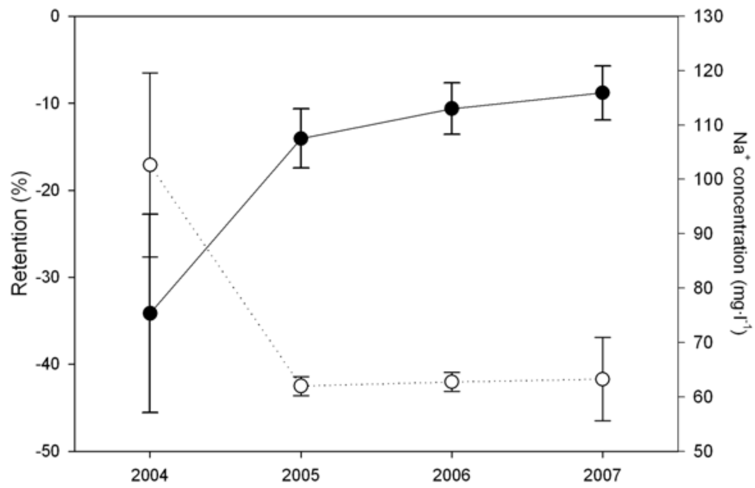

Figure 4. Temporal evolution of Na^+ retention rate and inflow concentration along the study period for the S-M-L-wetlands. Symbols represent mean±SE.

The NO_3-N concentration at the inflow throughout the study was homogeneous (19.90±0.48 mg·l^{-1}; Fig. 5). In this analysis, the data from the H-wetlands were separated because of the high variability and the smaller sample size that resulted from the extreme hydrological conditions. In 2006, the average NO_3-N concentration at the inflow rose to ~23 mg·l^{-1}. Overall, the retention rates of NO_3-N among all of the wetlands increased from ~40% in 2004 to ~50% in 2005, declined to ~25% in 2006 before increasing to ~50% in 2007. At the end of 2007, the retention rate reached 60%. In 2004, the retention rates of Na^+ were highly variable. From 2005 to 2007, the Na^+ concentration at the inflow varied little (62.25±1.83 mg·l^{-1}; Fig. 5). In the S-M-L-wetlands, the release of Na^+ to outflow water was progressively reduced from 2004 to 2007 (Fig. 5).

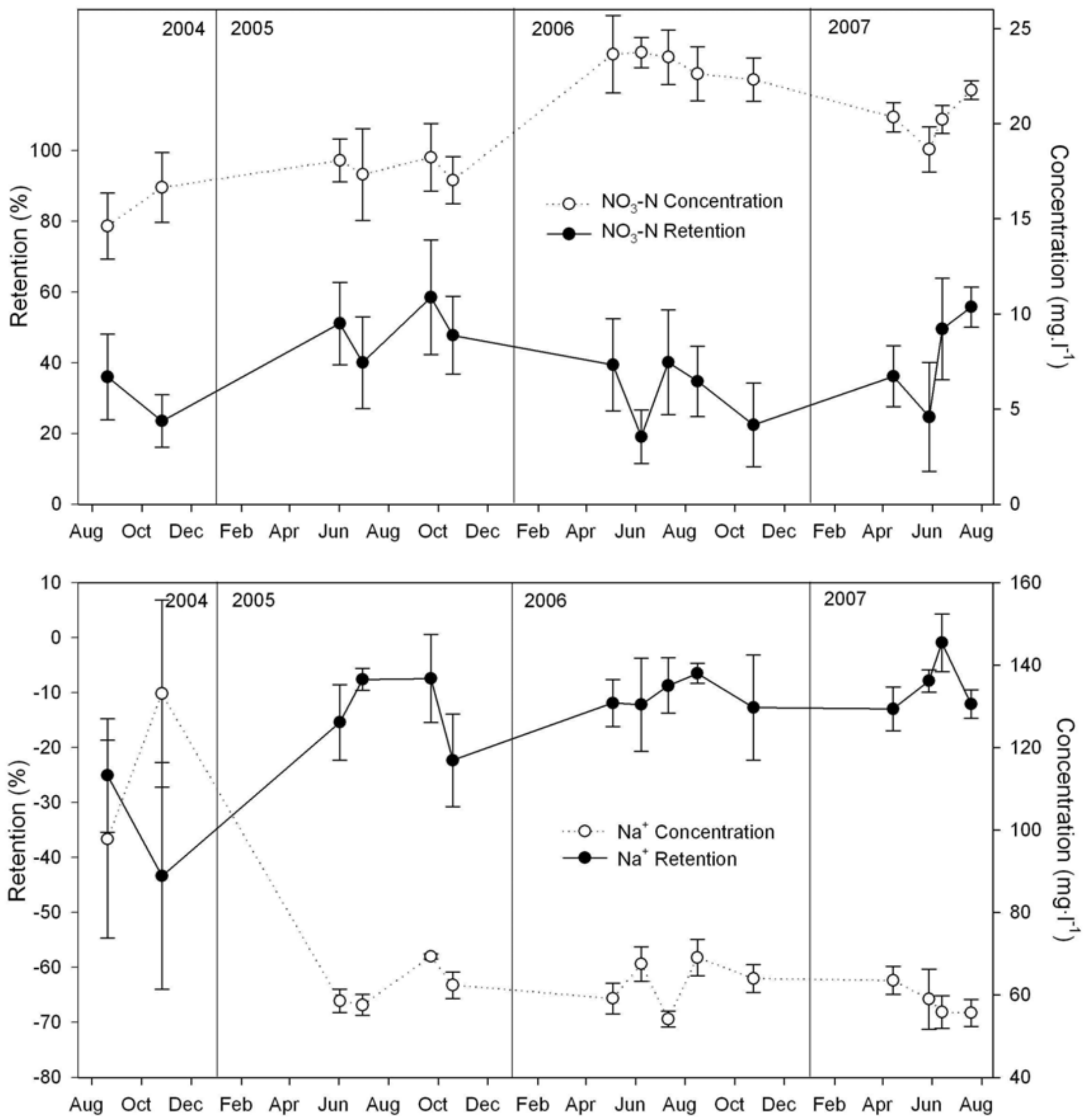

Figure 5. Temporal evolution of the retention and inflow concentration of Na^+ and NO_3-N along the complete study period for all wetlands. Symbols represent mean±SE.

Discussion

Nutrients

Surface flow wetlands constructed in old agricultural fields showed significant differences between inflow and outflow water for most of the measured parameters after spontaneous revegetation. All of the wetlands in the study appeared to be very efficient in NO_3-N retention, both by volume and mass. In the S (50 m^2), M (200 m^2), and L (800 m^2) wetlands, NO_3-N retention increased from ~30% in 2004 to >50% in 2007. N retention appeared to be unaffected by the size of the wetland. No significant differences for nitrogen removal were found between wetlands by size. N-retention was significantly higher in the H wetlands (5000 m^2) than it was in the smaller wetlands and, throughout 2007; those large wetlands eliminated almost all of the NO_3-N in the wastewater (~99%). N-retention by mass was almost the same for all sizes all along the year (50 and 90 $gN \cdot m^{-2}$). It means that wetland size should be closer to 5000 m^2 than to 800 m^2 to remove N from water. The 99% N-retention rates observed in this study are similar to those recorded in ~1-ha wetlands in the Ebro River Delta (Comín et al., 2001). Retention rates of NO_3-N in the in S-M-L-wetlands were similar to those observed in wetlands on forest soils in Midwestern USA (30-40%; Kovacic et al., 2006). Fink and Mitsch (2004) observed a retention of about 40 $gN \cdot m^{-2}$ in a 1.2-ha SFW in Ohio, USA, during storm events. In cold climates, retentions of 50-285 $gN \cdot m^{-2}$ were recorded in small (<1000 m^2) wetlands in Norway (Braskerud, 2002b) and, in similarly sized wetlands in Finland, rates were 6 to 45 $gN \cdot m^{-2}$ (Koskiaho et al., 2003).

In our study, four years after the wetlands were constructed, P-retention had not been achieved. The high accumulation of P within the soil matrix throughout the period in which the land was used for agriculture (20.50±6.41 $\mu g \cdot g^{-1}$ of available P —mean±SD—; Moreno, unpublished data) meant that most of the soil particles that were capable of retaining P were already occupied when the abandoned croplands were flooded. After the soils were flooded, some of the available P was released into the water column and exported. That process is enhanced by the drying and re-wetting conditions that occur during the driest periods (Venterink et al., 2002). During the four years of our study, P-release did not decrease significantly. H wetlands tended to export more TP than did the S-M-L-wetlands (Fig. 3). H-wetlands retained Ca significantly, probably because of the capacity of Ca to interact with the large amount of available P that was released by the soil into the water column. Consequently, the Ca-P system controlled the Ca and P concentrations in the outflow water, which indicated that some of the released P and dissolved Ca were fixed by the precipitation of Ca-P minerals (Shenker et al., 2005). In North Carolina, P-release to the flowing water occurred six years after the construction of wetlands on abandoned agricultural fields (Bruland et al., 2003).

Salts

Na-release and, to a lesser extent, Mg-release, clearly decreased over the four years that followed the construction of the wetlands (Fig. 4). Na-release changed from 100-500 to 70-250 $gNa \cdot m^{-2}$ per year. Although the decline in Na-release appeared to be asymptotic (Fig. 4), we predict that Na-release will be close to nil in a few years. The high release of Na is due to the high salinity of the soil matrix (226.41±232.90 $mg \cdot l^{-1}$; Moreno, unpublished data). In 2005, all of the cations (Ca^{2+}, Mg^{2+}, Na^{+} and K^{+}) had significant releases into the water outflow. In that year, a severe drought necessitated leaving the wetlands without water for periods, but later they were rewetted. Drying and rewetting of saline mudstones can promote the capillary ascent of additional salts from the deeper layers of the soil (Rodríguez-Ochoa et al., 1998). The high EC recorded at the outflow of the H-wetlands might have been due to the longer time that the water was in contact with the soil and the larger in surface, which provided larger amounts of soluble salts.

Other parameters

The increase in the alkalinity (i.e., the concentration of $CaCO_3$) at the outflow probably was caused by two processes. The main process was the release of $CaCO_3$ from the soil matrix, and the second process was the release caused by microbial activity during respiration (McCartney et al., 2003). The increase in carbonates in the water probably was the cause of the small increase in the water pH. Intermittent flooding and the rewetting of soils could favour the decalcification of soils because of the increase in CO_3^{2-} pressure and the drainage of pore-water solutes (Van den Berg and Loch, 2000), which will be greater in H-wetlands where water-soil contact is greater and might explain the four-fold difference in the concentrations of $CaCO_3$ at outflows than at inflows observed in 2007. Small amounts of lime can enhance the retention of P in the soil matrix (Zurayk et al., 1997). We hypothesize suspect that the system soil matrix-water column was entirely saturated with $CaCO_3$ and P, and P-retention by carbonated soils was not functional at this stage in the H wetlands. The release of CO_3^{2-} suggests a leak of C from the soil of the system into the environment.

Water Residence Time

In the H wetlands, at the time of maximum N-retention (~99% in 2007), WRT varied from 4 to 14 d. In similar constructed wetlands, retention rates of 99% and WRT of 5 d have been recorded (Comín et al., 2001). In boreal regions, Koskiaho et al. (2003) found that the highest retention rates of TP and TN occurred in the wetlands that had the longest WRT (1.63 d). Mitsch and Gosselink (2000a) reported an optimum range of 5-14 d of WRT for nutrient removal. In H-wetlands, a WRT of 4 d could satisfy the conditions for N-retention. The WRT was longer (1-2 d) in the H wetlands than it was in the S-M-L-wetlands (Table 1), where it was insufficient for complete N-retention. On the other hand, the longer the WRT the higher the concentrations of TP, salts, and sulphates at the outflow.

Temporal evolution

After the retention rate of NO_3-N by volume increased from 2004 to 2005, it dropped to the half in 2006. In this year, the NO_3-N concentration at the inflow increased because of an increase in the amount of fertilizer applied to irrigated fields upstream (Pujol, personal communication). The retention rate in 2006 of the wetlands by mass was almost the same than in 2005; consequently, the retention rate by volume decreased, which meant that in those years the size of the wetlands was insufficient to retain N. In 2007, the retention rate by volume increased to the 2005 level because of a reduction in the amount of fertilizer used in that year and, consequently, a decrease in the amount of N at the inflow. In the second half of 2007, all of the wetlands exhibited increases in their retention rates, which were >60%, which suggests progress towards general wetland stabilization in N-retention.

Throughout the study, TP levels were highly variable and no temporal trends were apparent. Despite a constant concentration of Na^+ at the inflow after 2004, the Na^+-release tended to be reduced. That trend also suggests and approach toward the stabilization of the wetlands after their construction on abandoned, salinized, agricultural soils.

Catchment considerations

Considering the nitrogen retention rates (50-90 $mg \cdot m^{-2}$ per year, by mass) showed here, and the annual exportation of NO_3-N in the study catchment where wetlands were located of ~730 kgN, a percentage of 3.64 and 6.54% of the catchment will be necessary to use as wetlands to ensure an optimal removal of N from the water flown into the hydric network. In an earlier study of those wetlands, we suggested that 3.25-5.60% of the catchment should be left as wetlands (Moreno et al., 2007), but those values were based on retention rates by volume. This small difference is caused by the method of calculation used, which was using retention by volume in Moreno et al. (2007) and by mass now. As the later one is more restrictive, it requires a larger area, this one is considered better to ensure efficient N removal in the wetlands.

Larson et al. (2000) suggests using a 22:1 ratio for wetlands in small agricultural catchments in Illinois, which is equivalent to 4.7% of the catchment, and Kovacic et al. (2006) suggested 5% in small agricultural catchments in Midwestern USA. In the Mississippi River Basin, Mitsch et al. (2001) advised reserving 3- 5% of the catchment for wetland restoration. Currently, wetlands used in our study are 5.77% of the catchment area, which is within in the correct range of the wetland/catchment rate, but they did not work optimally. Thus, to ensure water quality improvement, not only is enough having a wetland/catchment rate within the adequate range. Other requirements (e.g., biological structure, morphological design, soil characteristics) to achieve the challenge of improving efficiently agricultural wastewater discharged in the hydric network using constructed wetlands.

Conclusions and recommendations

In Monegros, the wetlands had not stabilized four years after they were constructed, and P, salts and, to a lesser extent, carbonates were being exported. By the end of the study, the release of salts was close to nil (only 9% in 2007). The characteristics of the soils (calcareous, agricultural, and salinized) had stronger effects on the characteristics (especially P, salts, and carbonates) of the outflow water than did the inflow wastewater. It could be possible a reduction in the release of P and C, particularly by reducing WRT. In dry years, intermittent flooding appeared to have an important influence on the amount of salts released. Permanent flooding can minimize that risk.

In all of the wetlands, important retention rates of N by mass were achieved (50-90 $gN \cdot m^{-2}$), but only the largest (5000 m^2) wetlands achieved optimal N-retention (99%); however, large wetlands had longer WRT and exported TP, salts, and C in larger amounts than did small wetlands. We conclude that individual wetlands should be closer to 5000 m^2 than to 800 m^2, and have WRT of about 4 d. Wetlands of that sizes should cover 3.5-6.5% of the catchment.

The incorporation of created wetlands into new agricultural developments or in those already existent will allow land planners to improve the water quality in irrigated agricultural catchments of in the semi-arid region of the Ebro River Basin. Land planners must recognize that the agricultural soils of Monegros and the Ebro River Basin have problems with over fertilization and salinization, which can delay the development of optimal water quality. With that in mind, we strongly recommend the use of wetlands to improve water quality that also can provide ancillary benefits (landscape heterogeneity, biodiversity reinforcement or soil improvement) in areas heavily degraded by intensive agriculture.

Acknowledgements

This research was supported by CICYT (REN2003-03040) and the Aragon Government–D.G. Investigación, Innovación y Tecnología (PMO19/2006), Instituto de Estudios Altoragoneses (Diputación Provincial de Huesca) and Research Group on Ecological Restoration (E 61).

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Chapter 4

Effects of wetland construction on nutrients, SOM and salts contents in semi-arid zones degraded by intensive agricultural use*

Abstract

Agricultural irrigation has contributed to the salinization/sodification of a significant amount of the soils in the Ebro River Basin (NE Spain). The aim of this study was to determine whether created wetlands affect soils by altering the levels of soil organic matter (SOM), inorganic nutrients contributed by fertilization and soil salinity/sodicity. We compared the properties of soils in permanently and intermittently flooded wetlands artificially created in conventionally tilled agricultural areas seven years after abandonment. To assess the long-term effects of wetlands, 18 wetlands created spontaneously by flooding with water from irrigated fields were compared to nearby croplands. Artificial wetlands did not exhibit a significant increase in SOM, but soil salinity was reduced, and inorganic nutrients were significantly lower than they were in the reduced in regard to reference croplands. In the soils of the spontaneous wetlands that formed after irrigation, SOM increased significantly, and the levels of inorganic nutrients and salinity were reduced compared to agricultural soils. Spontaneous wetlands appeared after irrigation showed a significant increase in SOM and similar reductions in inorganic nutrients and salinity. In the Ebro river Basin, seven years of restoration might not be sufficient time for a significant accumulation of SOM but, after a prolonged period (~40 yr), it increased. In constructed wetlands, inorganic nutrients were slowly converted into SOM, but a portion of the NO_3-N was denitrified, and some of the P was adsorbed by soil particles. Guidelines are provided for the creation of wetlands in existing and new agricultural developments in semi-arid regions.

Resumen

La agricultura en regadío ha contribuido a la salinización/sodificación de una cantidad significativa de suelos en la cuenca del Ebro (NE España). El objetivo de este estudio fue determinar como los humedales creados afectan a los suelos alterando los niveles de materia orgánica, los nutrientes inorgánicos aportados por la fertilización y la salinidad/sodicidad. Comparamos las propiedades del suelo en humedales permanente e intermitentemente inundados creados de forma artificial en suelos agrícolas labrados de forma convencional y abandonados siete años antes. Para valorar el efecto a largo plazo de los humedales, se compararon los suelos de 18 humedales crecidos

* Moreno, D., Comín, F.A., Pedrocchi, C. and Rodríguez-Ochoa, R. 2008. *Applied soil ecology*. In press.

espontáneamente a raíz de la transformación en regadío con suelos agrícolas adyacentes. Los humedales artificiales no mostraron un aumento significativo de materia orgánica, pero la salinidad del suelo se redujo, y los nutrientes inorgánicos fueron significativamente menores de lo que fueron en los suelos agrícolas. Los humedales espontáneos aparecidos tras el regadío mostraron un aumento significativo de materia orgánica y reducciones similares de nutrientes inorgánicos y salinidad. En la cuenca del Ebro, siete años de restauración podría no ser suficiente tiempo para una acumulación significativa de materia orgánica pero, tras prolongados periodos (~40 años) se incrementa. En los humedales construidos, los nutrientes inorgánicos fueron lentamente convertidos en materia orgánica, pero una parte del nitrato fue desnitrificada, y parte del fósforo fue adsorbido por las partículas del suelo. Algunas directrices son aportadas para la creación de humedales en desarrollos agrícolas existentes y futuros en regiones semiáridas.

Introduction

Inundation of abandoned agricultural zones is a common practice to create and restore wetlands and also as means for improving water quality, flood attenuation, rehabilitating degraded areas, and increasing habitat for wildlife (Anderson et al., 2005). Typically, abandoned agricultural soils have been submitted to practices leading to nutrient enrichment, loss of organic matter, compaction, and disruption of the soil profile (Bruland et al., 2003; Zedler, 2003). The creation and restoration of wetlands by flooding these types of zones fosters biogeochemical functions that improve soil properties in the short, medium, and long term (Zedler, 2000).

Numerous studies have evaluated the conditions of the soils in created and restored wetlands, often by comparing them to natural wetlands (Zedler and Callaway, 1999; Campbell et al., 2002; Craft et al., 2002; Bruland and Richardson, 2005b; Lu et al., 2007). Restored or created wetlands are viewed as preliminary states that have few operative ecological functions, but advance toward more stable states characterize by relatively high soil organic matter (SOM), total organic carbon (TOC), spatial variability, and low bulk density (D_B). Some studies have illustrated a general improvement of old agricultural soils when converted to wetlands. They have illustrated an increment of SOM and TOC (Hogan et al., 2004; Bruland and Richardson, 2006), a reduction in D_B and pH (Nair et al., 2001; Bruland et al., 2003), and an initial release of P from accumulated manure (Van Dijk et al., 2004; Aldous et al., 2005), which can be reversed after persistent flooding (Hogan et al., 2004). SOM is a key indicator of soil quality because it influences important soil processes such as respiration, denitrification and P-sorption (Williamson and Johnson, 1994; Bruland and Richardson, 2004; Bruland and Richardson, 2006).

Relatively little is know about the effects of wetland restoration on the condition of saline or salinized/sodificated soils. In semi-arid regions, agricultural irrigation is the main cause of the salinization of wetland soils, which is a major concern globally (Boettinger and Richardson, 2001; Tedeschi et al., 2001a; Houk et al., 2006). High salinity (~20 mS cm^{-1}) can reduce the redoximorphic features of wet soils (Boettinger and Richardson, 2001). In restored wetlands (e.g., wastewater treatment wetlands), the concentrations of cations (especially Ca and Mg) can increase significantly because of water circulation, which can lead to an increase in soil pH and affect P-deposition (Bruland et al., 2003; Anderson et al., 2005). There are also evidences of the significant reduction of nutrients in water passing through agricultural zones restored as wetlands and used for agricultural wastewater treatment as compared to agricultural non-restored sites (Comín et al., 2001; Bruland et al., 2003).

The objective of this paper is to assess the effect of the presence of permanent and intermittently flooded wetlands established in degraded agricultural soils on their primary soil characteristics. We hypothesized that wetlands improve soils by (1) increasing soil organic matter, which is depleted by intensive agricultural practices, (2) reducing inorganic nutrients, especially nitrate and phosphate, which accumulate in soils because of chronic fertilization, and (3) reducing soil salinity, which increases in the top layers because of chronic watering; during dry periods, salts from deeper horizons ascend the soil by capillarity. Given the high productivity of wetlands compared to that of typical abandoned fields, if the hypothesis is supported, created wetlands could be used by land managers as a tool to establish strategies to restore soils and wetlands within irrigated agricultural developments and modernizations at the catchment scale in a reasonable period. Also, the soils of the created wetlands will accumulate TOC and, subsequently, increase the sequestration of C from the environment.

Study area

Monegros is a 270 000-ha, inland, semi-arid, Mediterranean region (average annual temperature = 14.5ºC, average annual precipitation = 400 mm) in the centre of the Ebro River Basin, NE Spain (Fig. 1), that has high interannual variability (Comín and Williams, 1993; Comín et al., 2001; Moreno et al., 2007). The severely altered landscape consists of small plateaus interspersed with valleys, some gentle hills, and low-elevation mountain ranges. In the second half of the 20th Century, most of the land was converted into irrigated agricultural fields, particularly, of maize, alfalfa, and cereals. Today, much of the soil is salinized and sodified, and field abandonment is widespread in many parts of the region. As a result, the original pseudo-steppe vegetation, predominantly rosemary (*Rosmarinus officinalis*), thyme (*Thymus* sp.), halophytes (*Salicornia* sp.), and perennial grasses (*Lygeum spartium*, *Brachypodium retusum*), has been modified into a landscape dominated by halophytic species (*Tamarix africana*, *Atriplex halimus*, *Suaeda vera*) in abandoned, dry elevated zones, and by salt tolerant helophytes (*Phragmites australis*, *Typha latifolia*, *Scirpus holoschoenus*, *S. maritimus*, *Carex divisa*) in the wetter and lower zones of the valleys, where wastewater from irrigated fields is discharged.

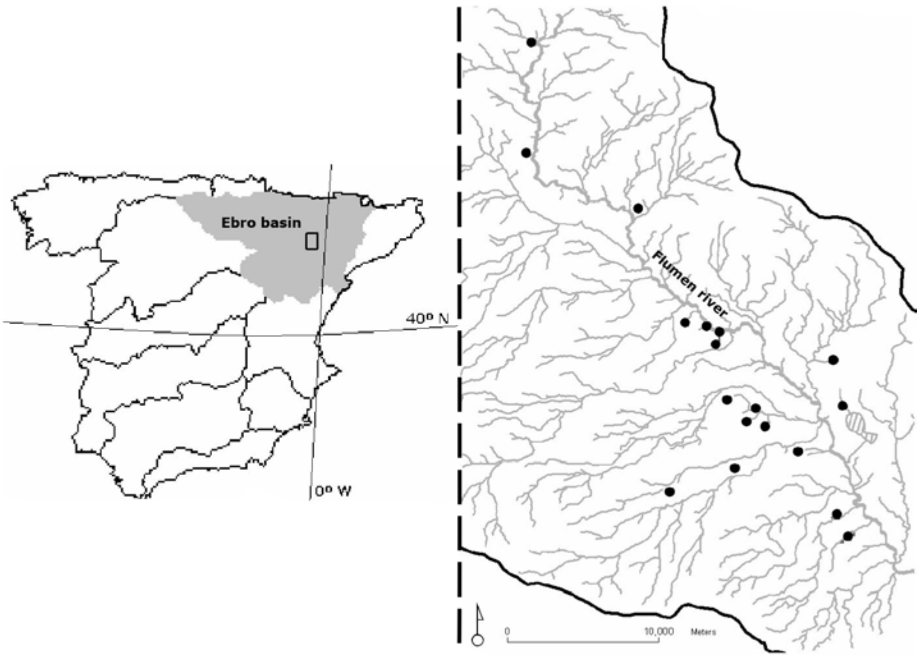

Figure 1. Location map of the study area (insert) with the main catchments of the Spanish Iberian Peninsula (left). Experimental ponds were located in the square. Because selected wetlands and croplands are small their locations are highlighted as dots. Wetlands and croplands were included in the Flumen basin and its drainage network (right).

Most of the wetlands that have permanent or intermittent standing water have stands of *Phragmites australis,* which are increasing their influence on plant communities and spreading because of their tolerance to fluctuations in water levels and salinity (Lissner et al., 1999a). The soils are mostly a Tertiary structure consisting of mudstones, with different degrees of salinity (conductivity ranges between 1-20 mS cm^{-1}), and inserted sandstone strati. Upper horizons were affected by soil salinization/sodification in two ways: induction by non-topsoil preservation during land levelling and anthropogenic changes in hydrological conditions. Those changes permitted salts to rise from the parent material (saline mudstones) into the upper horizons, especially when the soil is drying after a recent watering (Rodríguez-Ochoa et al., 1998). The soils of the study area were classified as Xeric Torriorthents (fine, coarse-to-fine loamy, mixed, calcareous, thermic, semi-active) under hydric conditions.

In the study area, experimental wetland ponds and reference croplands in five, previously irrigated, agricultural fields were studied (0.5 ha each one; Fig. 2). Three fields were abandoned for six years and contained wetlands, and two fields followed conventional agricultural tillage. To study the effect of permanent and intermittent flooding on soils, wetlands were flooded with agricultural wastewater from soil infiltration of adjacent irrigated fields. One field contained nine wetlands that were permanently flooded (PF), two fields were intermittently flooded (IF) when there was surplus water in the PF wetlands (because of restrictions on the water supplied to the adjacent croplands during the driest periods), and two active agricultural fields where left as references (AG). The fields containing wetlands were virtually covered by *Phragmites australis* (with scattered stands of *Typha latifolia, Scirpus lacustris,* and *Carex divisa*), which had grown naturally since the fields were abandoned, and croplands were sowed with alfalfa or left fallow.

To assess the soils in areas that were permanently flooded over long periods, samples were collected from 18 natural wetlands and 18 adjacent (within 50 m) croplands in an 80 000-ha area of Monegros (Fig. 1). In those areas, all of the wetlands that were larger than 1.0 ha (n=8) were included in the study (the largest was ~6.5 ha). Smaller (0.1 – 1.0 ha) wetlands (n=10) were included if they were topographically similar to the larger wetlands, similar catchment area, had homogeneous reed cover, and were accessible. All of the wetlands and croplands were in agricultural catchments where irrigated fields covered most of the area. All of the wetlands, except one (Sangarrén), received agricultural wastewater from irrigation surpluses through drainage channels or via groundwater that flowed into the fluvial system. The hydrologic system was entirely dependent upon the irrigation system in the fields of the surrounding area; thus, water drained into the wetlands during the meteorologically driest periods. Following the conversion into irrigated farmlands, the plant communities developed naturally, except in Sangarrén, which was fed mainly by natural groundwater. The location of the wetlands ranged from hillsides that had wastewater outcrops to flat valley floors. All of the croplands were active and sowed with alfalfa, maize, cereals, or left fallow.

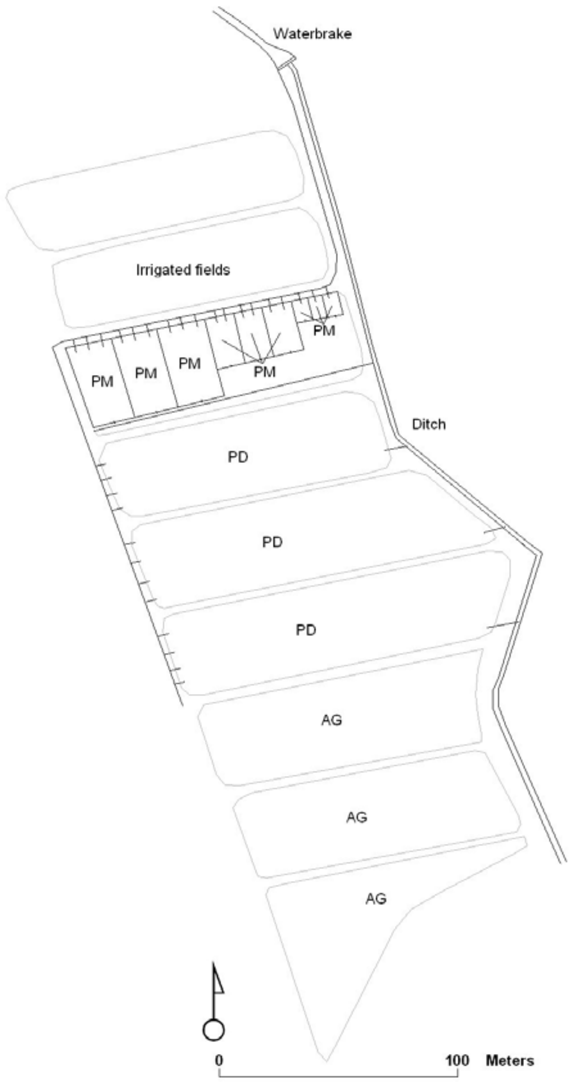

Figure 2. Plan of the experimental area. PM means permanently flooded ponds, PD means periodically flooded ponds and AG means agricultural reference fields.

Methods

Field sampling

Forty-two samples were collected in the study area, half of them in February and the other half in August 2006 (nine from PF ponds, six from IF ponds, and six from AG). This separation was carried out to estimate possible effects of seasonality on soil properties. In February, the sampling points were the geographic centroids of each pond or each of the three equal sections in which the fields were divided. In August, the sampling points were 1 m east of the sampling points in February. At each location, all of the aboveground biomass was collected within a 40 cm x 40 cm quadrat that surrounded the centroid. A 5-cm Ø plastic sleeve inserted into a steel piston was used to collect 30-cm soil cores. The sleeves were removed from the corer, stored in an ice-packed cooler, and transported to the laboratory. Samples were extracted from the sleeves and divided into two sections (0-10 cm and 10-30 cm).

In May 2006, soil samples were collected from the 18 spontaneous wetlands and the 18 adjacent croplands. Samples were collected at the geographical centroid of each wetland and, within croplands, at points that were 50 m away on the straight line that connected the geographical centroids of the wetlands and adjacent croplands. In the laboratory, those samples were homogenised manually, rather than divided into two sections.

Soil sample analyses

On the day they were collected, the sections of the samples were weighed before being air-dried, and soil parameters were measured (Table 1). To determine moisture content and D_B, the sections were reweighed after they reached a constant mass. A mortar and pestle were used to crush the samples and, to remove rock fragments and large roots and rhizomes, they were passed through a 2-mm sieve. The rest of soil parameters were measured in the subsamples of the fine-earth fraction obtained from each section. To estimate SOM, the Loss-on-Ignition method was used and the soil subsamples were placed into a muffle furnace at 450 ºC for 4 h. Soil texture (% clay, % silt, % sand) was quantified by the Discontinuous Sedimentation Method. Following Buurman *et al.* (1996), the Olsen-P Extraction Method and spectrophotometry were used to derive estimates of available P. The Wet Oxidation (Kjeldahl) Procedure quantified total N, and a 1 M KCl extraction followed by autoanalyzer provided estimates of NO_3^-N content. A Büchner funnel that had a hardened filter was used to extract a saturated paste, and a pH-meter and a conductimeter were used to measure pH and EC, respectively. To measure the concentrations of soluble Na, K, Mg, and Ca in the extract, we used ionic

Table 1. Mean±SD of all the soil properties measured in the experimental area and the existing wetlands for each treatment

Soil property	Abbreviation	Experimental area			Existing wetlands	
		Wetland	Dry wetland	Cropland	Wetland	Cropland
Phisical						
Moist content /%	MC	22.77±7.20	22.95±7.81	19.11±7.84	26.63±11.44	14.52±6.52
Clay /%		27.79±6.65	26.34±8.09	26.11±7.22	18.63±8.93	19.73±5.17
Silt /%		42.56±8.77	41.3±7.78	54.91±10.50	46.61±19.81	39.33±11.81
Sand /%		29.61±11.37	32.27±12.84	18.96±13.07	34.73±19.13	40.65±13.72
Soil organic matter /%	SOM	4.10±2.12	4.21±2.25	3.80±1.30	4.18±3.76	2.44±1.11
Bulk density /g cm^{-3}	D_b	1.43±0.25	1.40±0.24	1.39±0.28	1.21±0.22	1.38±0.26
Aboveground biomass /Kg m^{-2}	AB	2.98±1.50	1.93±0.74	0.00±0.00	3.34±1.76	0.00±0.00
Chemical						
pH		8.00±0.29	8.00±0.27	7.88±0.22	8.23±0.56	8.31±0.35
Electrical conductivity /dS cm^{-1}	EC	3.42±3.34	5.05±3.86	7.51±5.87	1.19±0.99	3.39±3.65
Redox /mV		-397.64±130.37	90.42±263.43	29.00±66.62	—	—
Available phosphorus /µg g^{-1}	Av. P	44.52±24.98	49.37±22.39	64.05±20.02	64.49±27.91	73.28±45.39
Total nitrogen /µg g^{-1}	TN	741.72±265.39	774.29±336.38	771.46±220.72	870.83±467.47	647.96±307.35
Nitrate nitrogen /µg g^{-1}	NO_3-N	15.25±21.25	24.83±28.63	37.35±53.61	58.84±100.37	185.26±206.10
Total organic carbon / mg g^{-1}	TOC	10.93±4.89	12.47±6.70	10.28±2.73	21.65±19.45	10.64±5.33
Carbon nitrogen ratio[a]	C/N	15.70±7.93	17.56±10.66	14.36±5.67	30.41±32.98	18.38±11.02
Soluble sodium /µg g^{-1}	Sol. Na	102.94±121.55	157.00±139.00	226.41±232.90	140.95±351.33	324.54±500.56
Soluble potassium /µg g^{-1}	Sol. K	1.70±1.66	3.42±4.44	3.75±1.03	4.44±8.45	3.62±16.73
Soluble magnesium /µg g^{-1}	Sol. Mg	11.22±9.05	52.69±50.14	20.53±13.36	39.79±33.54	50.54±34.94
Soluble calcium /µg g^{-1}	Sol. Ca	42.50±38.00	14.20±11.91	97.84±66.60	127.61±104.99	102.43±86.33
Sodium adsorption ratio[b]	SAR	3.63±3.35	4.93±2.66	5.30±4.53	2.63±5.98	7.96±11.63

[a] Measured as $C/N = \frac{Org.C}{TN}$

[b] Measured as $SAR = \frac{[Na]_e}{\left[\frac{Ca^{2+}+Mg^{2+}}{2}\right]_e^{1/2}}$

chromatography. To quantify organic C, the amount of inorganic C was subtracted from the amount of total C. To that end, pestle and mortar were used to grind two 1-g subsamples. One of them was previously ignited at 550ºC for 5.5 h, and both were measured by dry combustion in a carbon analyzer (Nelson and Sommers, 1996). To measure redox potential in the field, a portable redox electrode was inserted the soil. In croplands, the readings were taken immediately after the soil was wetted. To quantify the aboveground plant biomass in the quadrats, the samples were dried at 60ºC for 48 h before being weighed.

Statistical analyses

The experimental study design was unbalanced (nine PF, six IF, and six AG); therefore, a generalized linear model (GLM) was used to identify statistically significant differences in the soil properties of treatments. Before running the GLM, all of the variables were log-transformed, which insured that all of the variables met the assumptions of normality and homogeneity of variances. The presence of standing water at the sampling point at the time of sampling (PWS) was entered as a covariate because of the amount of variance absorbed from the model. When main effects were significant, differences between treatments were determined through orthogonal contrasts with PWS as the covariate.

To determine whether time of year had a significant effect on the properties of the soils, a second GLM analysis was performed. To determine whether there were significant relationships among the soil properties in the experimental ponds, a partial correlation analysis was performed, with the effects of the different treatments controlled. To normalize the distribution of the data, the values of the soil parameters were rank-transformed before the analysis.

To test for differences between the spontaneous wetlands and their cropland counterparts in the properties of their soils, the data were subjected to a multivariate analysis of variance (MANOVA). All of the variables were log-transformed and, consequently, met the assumptions of normality and homogenous variances. In addition, *t*-tests were used to determine whether wetland/cropland (permanent and intermittent wetlands combined at one level) had a significant effect on the variables that the GLM and MANOVA did not identify as significant and those excluded from the analyses because of non-normality and heterocedasticity. For the GLM and MANOVA, the level for statistical significance was set at $p<0.1$. For *t*-tests and the analyses of partial correlations, the value for statistical significance was set at $p<0.05$. All of the statistical analyses were performed with SPSS for Windows 13.0 (SPSS Inc., 2004).

Results

In the study area in Monegros, Spain, the soils in the experimental area and the spontaneous wetlands were in better condition than were those in the croplands, especially where standing water was present, which suggests that the presence of wetlands enhanced soil quality. In the experimental area, most of the soil parameters varied significantly (Table 2). EC was significantly reduced in a 73.57±3.52% (mean ±SE) (t=4.18; $p<0.001$) in the upper section of PF and 64.37±10.15% (t=2.00; $p<0.1$) in the lower section. TOC was significantly higher in the upper 10 cm of the soil samples from IF than they were in the corresponding samples from PF and AG (Table 2; Fig. 3). The TOC in IF was 39.22±4.37% higher than it was in the AG.

Soil property[a]	F	df	p
Experimental area			
0-10 cm			
Av. P	4.60	2	0.0163
NO_3-N	9.84	2	0.0004
TOC	4.20	2	0.0225
Sol. Na	11.63	2	0.0001
Sol. K	4.86	2	0.0132
Sol. Mg	6.17	2	0.0048
Sol. Ca	7.27	2	0.0021
SAR	8.72	2	0.0008
10-30 cm			
EC	7.16	2	0.0023
Av. P	4.68	2	0.0152
Sol. Na	4.27	2	0.0213
Sol. K	2.60	2	0.0873
Sol. Mg	3.53	2	0.0393
Sol. Ca	6.21	2	0.0046
Existing wetlands			
EC	5.35	1	0.0270
D_B	3.45	1	0.0718
SOM	4.73	1	0.0367
TN	3.23	1	0.0811
NO_3-N	9.40	1	0.0042
TOC	6.75	1	0.0138
Sol. Na	5.01	1	0.0318
SAR	8.16	1	0.0072

Table 2. Results of the GLM for the experimental area testing for significance of the effects of wetland, dry wetland and cropland treatments and ANOVA for the existing wetlands testing for significance of the effects of wetland and cropland treatments. Only significant results at a p<0.1 were showed.

[a] Definitions of abbreviations in Table 1.

Permanently flooded soils had significantly reduced NO_3-N (72.96±6.97%) in the upper 10 cm of the soil; however, differences in TN were not significant. At a depth of 10-30 cm, the concentrations of NO_3-N were very low. Significant reductions of phosphorus fractions occurred in both sections of the soil samples.

Available phosphorus diminished 36.82±5.83% and 34.17±13.03% in the upper and lower sections of the PF soil sample, respectively (Table 2; Fig. 3).

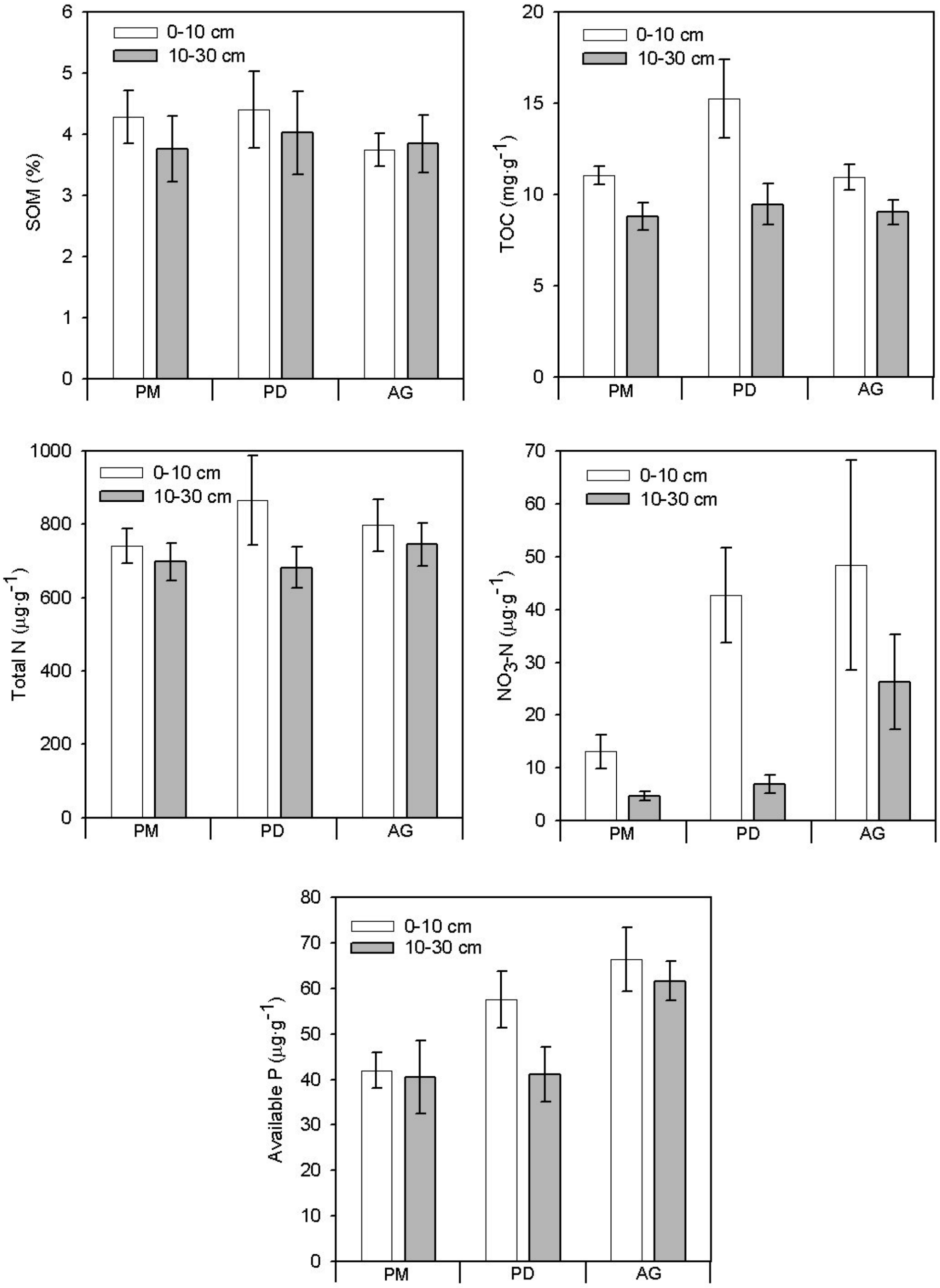

Figure 3. Variation of organic matter and nutrients related variables for the three treatments of the experimental area. Samples were divided in two subsamples (0-10 cm and 10-30 cm). PM means permanently flooded ponds, PD means periodically flooded ponds and AG means agricultural reference fields.

The concentrations of soluble cations significantly diminished in PF (Fig. 4). Soluble Na decreased by 76.66±6.71% and 65.34±3.35% in the upper and lower sections of the soil samples, respectively, and soluble Mg did it by 50.22±5.93% and 61.15±4.25%, respectively. Although soluble Ca was reduced significantly (57.13±42.95% and 72.16±10.72%) in the upper and lower sections of the soil samples from PF, the standard errors were high. In the samples from IF, the standard errors were even higher. Intermittently flooded wetlands exhibited significant decreasing of soluble K in regard to croplands (66.82±9.96% and 53.53±20.89% in the upper and lower sections of the soil samples, respectively), with high standard errors. In PF, the concentrations of soluble K were reduced (27.10±4.71% and 52.76±12.05% in the upper and lower sections of the soil samples, respectively) (upper section: F=4.86, p<0.05; lower section: F=2.60, p<0.1). In PF, SAR decreased by 62.88±5.46% and 35.36±7.46% in the upper and lower sections of the soil samples, respectively. In IF, the standard errors were large and, therefore, excluded from the analysis.

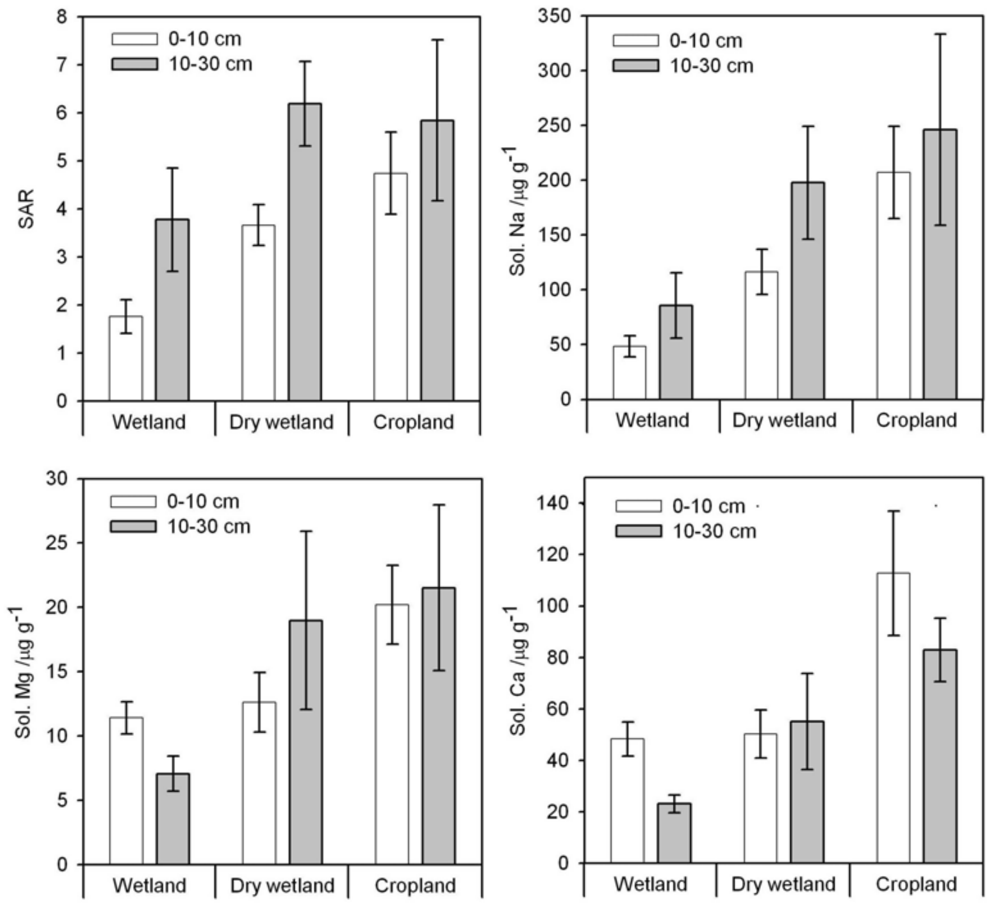

Figure 4. Variation of salinity related variables and main cations for the three treatments of the experimental area. Samples were divided in two subsamples (0-10 cm and 10-30 cm). PM means permanently flooded ponds, PD means periodically flooded ponds and AG means agricultural reference fields.

SOM and all nutrient variables (available P, TN, NO_3-N, and TOC) were positively correlated, especially in the upper 10 cm of the soil (Table 3).

Table 3. Partial rank correlations among the soil properties measured in the experimental area, controlling for the effect of different treatments (permanent wetlands, periodical wetlands and croplands). Upper and lower values of each row correspond to 0-10 and 10-30 cm depth respectively. Only those correlations significant at the *p*<0.05 level were showed (ns=not significant).[a]

	D_B	EC	Av. P	TN	NO_3-N	TOC	C/N	Sol. Na	Sol. K	Sol. Mg	Sol. Ca	SAR
SOM	0.469	ns	0.442	0.375	0.303	0.342	ns	ns	ns	ns	ns	ns
	0.693	ns	0.407	ns	ns	ns	ns	ns	ns	ns	ns	ns
D_B		ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
		ns	0.360	ns	ns	ns	ns	ns	ns	ns	ns	ns
EC			-0.521	ns	-0.322	ns	ns	0.773	0.493	0.740	0.482	0.584
			-0.321	ns	ns	ns	ns	0.851	0.404	0.712	0.560	0.705
Av. P				ns	0.546	0.337	ns	ns	ns	ns	ns	-0.369
				ns	ns	0.330	ns	ns	ns	ns	ns	-0.335
TN					ns	0.313	-0.561	ns	0.359	ns	ns	ns
					ns	ns	-0.412	ns	ns	ns	ns	ns
NO_3-N						ns	ns	ns	ns	-0.306	ns	ns
						ns	ns	ns	ns	ns	ns	ns
TOC							0.482	ns	ns	ns	ns	ns
							0.732	ns	ns	ns	ns	ns
C/N								ns	-0.322	ns	ns	ns
								ns	ns	ns	ns	ns
Sol. Na									0.353	0.718	0.390	0.840
									0.558	0.754	0.621	0.871
Sol. K										0.485	0.504	ns
										0.580	0.466	0.399
Sol. Mg											0.793	0.365
											0.891	0.448
Sol. Ca												ns
												ns

[a] Definitions of abbreviations in Table 1.

Negative correlations were found between nutrient parameters and salinity related variables and other cations. Available P was negatively correlated with EC, SAR and exch. Na (this in the lower 20 cm of the soil), NO_3-N was it with EC and exch. Mg (this in the upper 10 cm of the soil). In the upper 10 cm of the soil, soluble K and TN were positively correlated. SOM and BD were significantly and positively correlated in both sections of the soil samples. In the upper 10 cm of the soil, TOC was significantly positively correlated with available P and TN, and in the lower section, with available P. In the upper 10 cm of the soil, NO_3-N was significantly positively correlated with available P. Soil texture was significantly correlated with salinity-related variables (Table 4). Clay (%) was significantly positively correlated with all of the salinity-related variables, but soluble K and sand (%) were negatively correlated with most of these variables.

	EC	Na	Mg	Ca	SAR[a]
Clay /%	0.363	0.480	0.456	0.311	0.370
Silt /%	ns	ns	ns	ns	ns
Sand /%	-0.356	-0.472	ns	ns	-0.384

Table 4. Significant relations of the partial rank correlations among the soil texture and salinity related variables measured of the upper sections of sampling taken in the experimental area, controlling for the effect of different treatments (permanent wetlands, periodical wetlands and croplands). Only those correlations significant at the *p*<0.05 level were showed (ns=not significant).

[a] Definitions of abbreviations in Table 1.

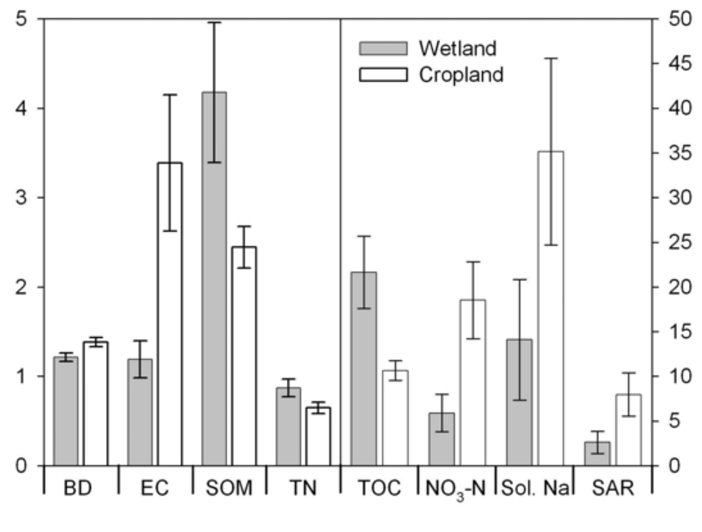

Figure 5. Variables defined as significant by the MANOVA test for difference between wetlands and adjacent croplands in the soils of valleys. Units are expressed as follows: D_B (gr cm^3), EC (mS cm^{-1}), SOM (%), TN (mg g^{-1}), TOC (µg g^{-1}), N-NO_3 (µg g^{-1} x 10^{-1}), Av. P (µg g^{-1}), Sol. Na (µg g^{-1} x 10^{-1}) and SAR (no units).

In the spontaneous wetlands similar reductions occurred, except in SOM and TOC (Table 2; Figure 5), which increased in wetlands in regard to croplands, but were highly variable among sites. SOM and TOC increased 70.85±20.90% and

103.46±30.61%, respectively. In the spontaneous wetlands, but not in the experimental area, BD was significantly lower (12.34±2.59%). In the spontaneous wetlands, EC and NO_3-N were reduced by 64.88±5.93% and 68.24±6.88%, respectively, which is similar to the reductions observed in the experimental area. Although TN was lower in wetlands than in croplands, the values of this parameter were highly variable (Fig. 4). The same was true for soluble Na and SAR in wetlands compared to croplands.

Discussion

The first hypothesis (increased SOM in wetland soils compared to agricultural fields) was not met in the created wetlands that had been abandoned for seven years, SOM was not significantly higher than it was in croplands, although values were higher at the PF and IF sites than at the AG sites. In the selected spontaneous wetlands (considered as the most likely way in which wetlands soils will evolve in Monegros), SOM (~70%) and TOC (~100%) were significantly higher than they were in adjacent croplands. Those increases occurred over the 40-50 years since the land was converted to irrigated agriculture and these wetlands were spontaneously created by wastewater from irrigated fields, although there was high variability among the sites sampled. The accumulation rates of SOM (0.03% per year) and TOC (0.22 mg g^{-1} per year) in the spontaneous wetlands of the study site were low compared those in created wetlands on mineral soils (0.5% of SOM per year, n = 10 yr) (Anderson et al., 2005), in reclaimed mines (12.5 mg g^{-1} of TOC per year, n = 16 yr) (Nair et al., 2001), and in reclaimed old rice paddies in China (1.2 mg·g^{-1} of TOC per year, n = 15 yr) (Lu et al., 2007). In Monegros, the slow rate of SOM accumulation might have been due to the effects of two processes that accelerated decomposition: the steady contribution of nutrients from agricultural wastewater (Craft, 2001), and the high temperatures during the vegetation growing period (Collins and Kuehl, 2001). In IF, the decomposition rate might be reduced during dry periods, which would allow greater accumulation in the upper 10 cm of the soil (Fig. 3). In Monegros, SOM and TOC were positively correlated with levels of soil nutrients (available P, TN, and NO_3-N), but negatively correlated with BD, which might have resulted from the protracted tilling of agricultural soils that prevented the positive effects of SOM accumulation on soil structure.

The second hypothesis (reduction of inorganic nutrients in wetland soils compared to agricultural fields) was fulfilled; after the wetlands were created, the inorganic nutrients from fertilization (NO_3-N and available P) that accumulated in the soil decreased substantially, even in the lower horizon (10-30 cm; Fig. 3). The TN did not decrease and was not correlated with NO_3-N, which indicates that most of TN was an organic fraction. In Monegros, the excessive fertilization of agricultural soils led to nutrient concentrations that were very high compared to those reported elsewhere (Bruland et al., 2003; Van Dijk et al., 2004), where NO_3-N concentrations in restored agricultural soils were reduced, because of plant growth and microbial activity (Silvan et al., 2003; Ibekwe et al., 2007). Reduction of NO_3-N in wetlands takes place by denitrification, among other processes, and its rate is dependent on the flooding regime (Hernandez and Mitsch, 2007). In our study, plant growth and microbial activity probably were responsible for the decrease in available P in the upper horizon (0-30 cm) because a net P release into the water column did not occur in the experimental wetlands. In small constructed wetlands in Finland under acidic conditions 25% pf P was immobilized in the microbial biomass (Silvan et al., 2003). P-sorption did not produce a net reduction in P from the water entering the wetlands as illustrated in previous studies (Moreno et al., 2007). It is likely that the P released into the water column compensated for the P adsorbed by the high levels of $CaCO_3$ and clays in the soils. A similar situation happened in restored wetlands of Israel where high concentration of Ca allows the Ca-P system controlling P concentration trough the precipitation of Ca-P minerals (Shenker et al., 2005). Bruland *et al.* (2003) reported a similar decrease in the concentrations of P in restored wetlands on old agricultural soils. In calcareous soils P-sorption has been also

In the spontaneous wetlands, the behaviour of N was similar to that of P. TN was significantly higher in these wetland soils than it was in the croplands, although inter-site variability was high, which suggests N accumulated in organic forms. The strong negative correlations between available P and soil parameters related to saline conditions (EC and SAR) could be the consequence of a retrogradation process of the precipitated organic P forms and prevented mineralization. The correlations between NO_3-N and EC were not as high, which reflected the depressing effect of salinity on nitrification (Irshad et al., 2005).

Under the conditions of our study area (nutrient concentrations of entering water, temperature and soil salinity), wetlands remove inorganic nutrients from their previous agricultural soils and organic nutrients accumulation is attenuated by the presence of saline conditions.

The third hypothesis (reduction of soil salinity in wetland soils compared to agricultural fields) was fulfilled. The EC, SAR, and the concentrations of soluble Na, Mg, and Ca were reduced significantly in the wetland soils, especially when PF and AG were compared. In IF wetlands, the values were between those of PF and AG, and were highly variable. The concentrations of Na, Mg and EC at the inflow and the outflow of the wetlands differ significantly after four years of functioning (Moreno, unpublished data), which indicates that the soluble cations were washed from the soil and released to the hydric network. The deposition of Ca and Mg as carbonates and bicarbonates did not occur in significant amounts because the inflow water had low concentrations of these cations; despite of it, small amounts of $CaCO_3$ and $MgCO_3$ were deposited in the upper horizons of the soils at the PF sites. However, decalcification processes of soils will be reinforced by the increase of CO_2 pressure occurred after inundation which is favoured by the high pH (>7.5) of these soils (Van den Berg and Loch, 2000). This partially avoids the accumulation of Ca in the soil. In IF and PF, the results were very similar, which suggests that flooding regime had a limited effect. At sites with intermittent flooding, the concentrations of soluble Ca and Mg were reduced, significantly. In this situation, aerobic conditions allows the oxidation of iron sulphides that also favour decalcification, and is added to the decalcification produced by the increase of CO_2 pressure during the anaerobic conditions (Van den Berg and Loch, 2000). The partial adsorption of soluble Ca, Mg and Na within the organic compounds incorporated to the soil might have favoured a decrease in their concentrations. Low-saline wastewater might have forestalled the salinization of the soils (Moreno et al., 2007). The high values of SAR in the 10-30 cm section of the soils at the PF sites, which were even higher than those recorded in the AG soils, resulted from the capillary movement of soluble cations from the deeper horizons (mudstones), which were rich in salts, as the soil dried following inundation (Rodríguez-Ochoa et al., 1998). Those soluble cations (mainly Na) were more readily available for adsorption by the clays of the upper horizons, which demonstrate the problem of allowing soils to dry intermittently. The strong correlations between clay content levels and the variables associated with salinity and cation concentrations support that interpretation. Furthermore, it suggests that clayey materials might pose problems for restoration activities when salinity has to be reduced.

The results of our study offer some guidelines for the reclamation of degraded and salinized soils. In the semi-arid and arid zones of the Ebro River Basin and similar environments, at least periods longer than seven years are needed for the development of SOM (because of the high decomposition rate) and an improvement in soil texture. The inorganic nutrients that accumulate in the soils because of agricultural practices and nutrient inflow from wastewater are converted into plant biomass and, subsequently, into SOM. Thus, creating permanently flooded wetlands using wastewater from irrigated fields is an efficient tool to restore or improve salinized soils. However, as it may take a few years for newly created wetlands to develop most of their structural and functional characteristics, we suggest establishing the new wetlands as a mosaic in the territory. This will allow for different wetlands to develop their structural and functional characteristics at different rates. This will also allow applying different management techniques (e.g., rotational plant harvesting, changing hydroperiods) in case other objectives are required also for these ecosystems together improving soil status (e.g., maximizing biodiversity, control mosquitoes), as it is the case while considering restoration at landscape scale (Comín et al., 2001; Moreno et al., 2007).

Incorporating created wetlands into new and old agricultural developments will allow land planners to recuperate soils that have been degraded by intensive agriculture and complement the more traditional practices of using wetlands to improve water quality or regulate flooding. Soil recuperation will allow new possibilities for soft agricultural and ranching exploitation in areas that previously were not suitable and will contribute to an increase in C sequestration in large potions of the semi-arid Mediterranean region that are used for irrigated agriculture.

Acknowledgements

This work was supported by CICYT (REN2003-03040) and Aragon Government–D.G. Investigación, Innovación y Tecnología (PMO19/2006) and Research Group on Ecological Restoration (E 61). Thanks are given to A. Rodríguez Vila for her collaboration in field and lab work.

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Chapter 5

Avian communities' preferences in recently created agricultural wetlands in irrigated landscapes*

Abstract

Numerous wetlands have been created spontaneously in the Ebro river basin as a consequence of new irrigation developments over the last 50 years. Water used for irrigating farmland drains into the lower parts of small valleys to form wetlands that are mostly dominated by common reeds (*Phragmites australis*). Bird communities established in these wetlands are still simple, partly due to the lack of management to enable their ecological functions to improve. A knowledge of which environmental features favour these bird communities is essential in order to improve the design of newly created or restored wetlands associated to future irrigation developments. The habitat and vegetation features of fifteen wetlands have been sampled. The structure of bird communities (richness, abundance and diversity) established in them has been monitored over three years during the breeding season and in winter — during daytime foraging and nocturnal roosting. The presence of bushes, height of stems and distance from large wetlands (>1 ha) proved to be the most influential variables on bird community structure and on most abundant species during the breeding season. Wetland area and compactness influenced species richness and the most abundant species during winter foraging and roosting. There was a maximum height at which only reed-dwelling birds remained. Uncontrolled winter burning had a severe negative effect upon these recently established populations. The ecological functions of newly constructed or restored wetlands, including those for wastewater treatment in agricultural catchments, could be substantially improved simply by considering a few guidelines during design and management.

Resumen

Numerosos humedales han aparecido espontáneamente en la cuenca del Ebro como consecuencia de los desarrollos agrícolas en regadío de los últimos 50 años. El agua utilizada para regar los campos agrícolas drena hacia las partes bajas de los valles y se forman humedales fundamentalmente dominados por carrizo (*Phragmites australis*). Las comunidades de aves establecidas en esos humedales son todavía simples, particularmente debido a la falta de gestión que permitiría que sus

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* Moreno, D., Pedrocchi, C. and Comín, F.A. *Biodiversity and Conservation*. Submitted.

funciones ecológicas mejorasen. El conocimiento de cuales son las características ambientales que favorecen las comunidades de aves es esencial con el fin de mejorar el diseño de nuevos humedales creados o restaurados asociados a futuros desarrollos agrícolas. Las características del hábitat y la vegetación de 15 humedales han sido muestreadas. La estructura de las comunidades de aves (riqueza, diversidad y abundancia) ha sido seguida durante tres años en la estación reproductora y la invernada — durante la alimentación diurna y como dormideros. La presencia de arbustos, altura de los tallos y la distancia a grandes humedales (>1 ha) fueron las variables más influyentes sobre la estructura de la comunidad de aves y sobre las especies más abundantes durante la estación reproductora. El área del humedal y la compacidad influyeron sobre la riqueza de especies y sobre las especies más abundantes durante la alimentación diurna y el descanso nocturno. Hubo una altura máxima a partir de la cual solo especialistas en juncos permanecían. Las quemas invernales incontroladas tuvieron un severo efecto negativo sobre estas recientes poblaciones. Las funciones ecológicas de los nuevos humedales construidos o restaurados, incluidos aquellos para el tratamiento de aguas residuales en cuencas agrícolas, podrían mejorar sustancialmente sólo con considerar unas pocas directrices durante su diseño y gestión.

Introduction

The transformation of natural landscapes and dry agricultural areas into irrigated farmland has led all around the world to the loss of natural wetlands which are important for birds, fisheries, plant communities and even human settlements (Lemly et al., 2000; Brinson and Malvarez, 2002; Czech and Parsons, 2002; Anderson et al., 2005; Galbrait et al., 2005). In other places, irrigation has led to an impoverishment of existing wetlands as a consequence of salinization, eutrophication, the alteration of hydrological systems or the destruction of part of them to create new irrigated fields and irrigation infrastructures (Carpenter et al., 1998; Lemly et al., 2000; Mañosa et al., 2001; Anderson et al., 2005; Moreno et al., 2008d). In most of Europe the area covered by wetlands dominated by the common reed (*Phragmites australis* (Cav.) Steudel) has suffered progressive dieback (Brix and Schierup, 1989; Van der Putten, 1997). However, the area covered by reedbeds has increased in extensive areas of semi-arid Spain after being transformed into irrigated land (Moreno et al., 2008d). On the other hand, irrigation is essential to ensure agricultural production in many arid and semi-arid regions of the world.

Some valuable benefits for bird communities can occur after transformation into irrigated farmland. In some places, these irrigated crops can be considered agricultural wetlands and this has enabled the establishment of bird communities that use them as foraging, roosting or breeding niches (Fasola and Ruiz, 1996; Lemly et al., 2000; Czech and Parsons, 2002). In other cases, wetlands appeared spontaneously due to the abandonment of fields, water storage or outcrops of agricultural leachates (Houlahan and Findlay, 2004; Sanchez-Zapata et al., 2005; Moreno et al., 2008d). Wetland construction or restoration in agricultural areas has been suggested as a valuable measure to palliate eutrophication, providing, at the same time, new habitats for the settlement of bird communities (Zedler, 2003; Anderson et al., 2005; Eades et al., 2005; Moreno et al., 2008d).

Human management is a common fact that determines the composition of animal and plant reedbed communities. Cutting, burning, grazing, flood control and the application of herbicides are some of the most common interventions on reedbeds that adversely affect bird communities when they are not controlled (Graveland, 1998; Martinez-Vilalta et al., 2002; Poulin and Lefebvre, 2002; Poulin et al., 2002; Eades et al., 2005). Winter burning is a common practice aimed at controlling reed expansion, improving cattle grazing, avoiding accidental fires, limiting invasive or woody species, or improving habitat quality for wintering waterfowl (Gabrey et al., 1999; Gabrey and Afton, 2004; Isacch et al., 2004; Brennan et al., 2005). In the Ebro valley these practices are not part of management plans, but reedbeds are systematically burned without a defined aim. The burning of reedbeds has often been recommended as a way to preserve a certain quality of habitat. It has been also suggested that burning an entire reedbed would have a devastating effect on wildlife (Hawke and José, 1996; Gabrey et al., 1999; Eades et al., 2005), that burning provides qualitative changes in communities of breeding birds (Gabrey and Afton, 2004) and that excessive burning may reduce the habitat available for endangered or rare species (Isacch et al., 2004).

Most reedbeds located in the Ebro valley are isolated patches within a matrix of irrigated fields. Several studies have reported on the effects that this patchiness can have on bird communities in other habitats (Gibbs, 1993; Diaz et al., 1998; Naugle et al., 1999; Horn et al., 2005; Davis et al., 2006; Huste et al., 2006; Scheffer et al., 2006), but very few have studied patchy reedbeds (Foppen et al., 2000; Surmacki, 2005; Paracuellos, 2006). These studies have detected the source-sink effects of bird species between patches, the influence of surrounding land uses on the abundance of birds and certain wetland features that facilitate the occupation of patches by birds. A few studies of large reedbeds (>10 ha) have addressed the question of how environmental variables affect the structure and composition of bird communities in Central Europe (Graveland, 1998; Baldi and Kisbenedek, 2000; Surmacki, 2005; Baldi, 2006; Trnka and Prokop, 2006), and the Mediterranean region (Martinez-Vilalta et al., 2002; Poulin et al., 2002). They have suggested that large sizes, few human interventions, diverse and natural vegetation structures, and scarcely altered hydrological systems favour a richness and abundance of birdlife. Baldi and Kisbenedek (1999; 2000) observed for the entire bird community of reedbeds that small patches had a higher relative species richness than large ones because of the effect of edges. This effect was also reported by Scheffer et al. (2006) for shallow lakes and ponds affected by isolation. All these studies focussed on bird communities established on naturally grown wetlands, most of which were solely considered passerines. What is needed is a study of complete bird communities that have recently been established in reedbeds. These communities are structurally simple and have low species richness that make reedbeds particularly suitable for testing bird-habitat relationships. It is also relevant that these wetlands were grown spontaneously as a consequence of outcrops of agricultural wastewater (Moreno et al., 2008d), with the only human intervention being uncontrolled burnings.

The objective of this study is to define the environmental characteristics of wetlands grown as consequence of agricultural runoff determining the structure of bird communities in an irrigated agricultural landscape of a semi-arid area of the Mediterranean region. Additionally, we wanted to know how our results might be affected by burning. This information will allow land planners to design wetlands that can facilitate the colonization and development of new bird communities and, therefore, to increase the biodiversity of agricultural landscapes that have been degraded by irrigation.

Study area

Monegros is a 270,000 ha inland semi-arid Mediterranean region (average annual temperature 14.5ºC, average annual precipitation 400 mm) located in the centre of the Ebro River basin, NE Spain (Fig. 1) with high interannual variability (Moreno et al., 2008d). Soils are mainly dominated by a Tertiary structure composed of clays with different salinization levels (conductivity ranges between 1-10 mS cm^{-1}) with inserted sandstone stratus. Landscape is severely altered and is composed of small plateaus with valleys in between, some gentle hills and low altitude mountain chains. Most of this land was transformed into irrigated agricultural fields from the 1950s to 1990s. Maize, alfalfa and cereal are the most common crops. Soil salinization and the abandonment of agriculture are now widespread in many parts of this region. As a consequence, the original pseudo-steppe vegetation, dominated by rosemary (*Rosmarinus officinalis*), thyme (*Thymus* sp.) and halophytes (*Salicornia* sp.) as well as perennial grasses (*Lygeum spartium*, *Brachypodium retusum*), was transformed into a landscape dominated by halophytic species (*Tamarix africana*, *Atriplex halimus*, *Suaeda vera*) in abandoned dry elevated zones and helophytes (*Phragmites australis*, *Typha latifolia*, *Scirpus holoschoenus*, *S. maritimus*, *Carex divisa*) in the abandoned wetter and lower zones of the valleys. In most of these zones, wetlands with permanent or intermittent surface water are formed and colonized by *Phragmites australis* which is spreading and increasingly dominating the plant community because of its tolerance to changing water levels and salinity (Lissner et al., 1999a).

Fifteen of these wetlands within an 80,000 ha zone of Monegros region were selected for this study (Fig. 1). All wetlands of this zone larger than one hectare (eight in all) were included (maximum size was ~6.5 ha). Additionally, other wetlands of sizes between 1000 m^2 and one hectare were selected in accordance with the following criteria: similar topographic and geographic situation to large wetlands, similar catchment area, homogeneity of reed cover and accessibility. Seven wetlands were selected according to this second selection. Sangarrén wetland presented exceptional characteristics because of its groundwater feeding and no similar wetlands

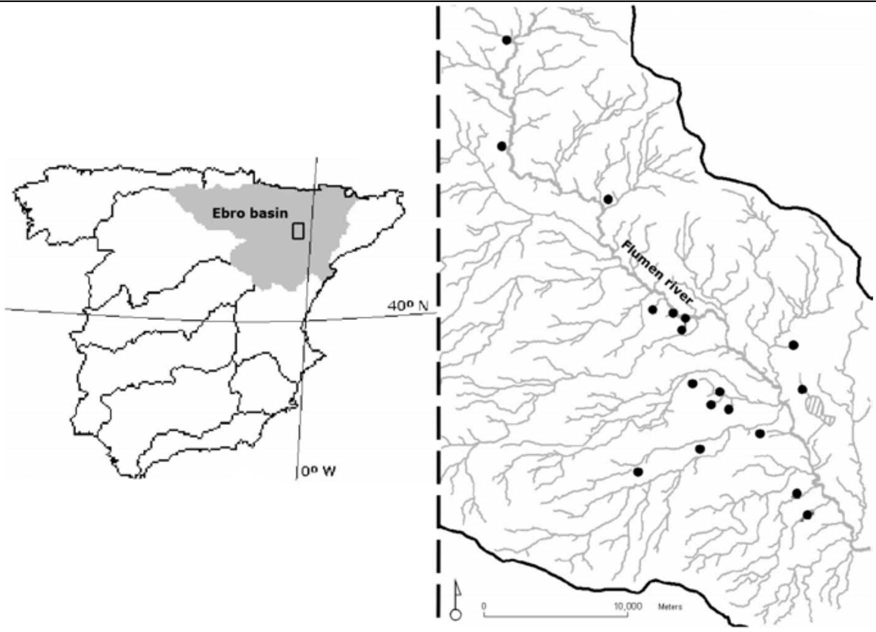

Figure 1. Location map of the study area (left) with the main catchments of the Spanish Iberian Peninsula (top). Because selected wetlands are small their locations are highlighted as dots. Wetlands are included in the Flumen basin and its drainage network (right).

of smaller sizes were found. All wetlands were located in agricultural catchments where irrigated fields occupied most of the surface. The wetlands received agricultural wastewater coming from irrigation surpluses through drainage channels or via groundwater and flowing into the fluvial system. The hydrologic system was entirely dependent on the system of irrigating fields in the surrounding area. Thus, water drained into the wetlands during the meteorologically driest period. The distribution of water around the wetland was irregular as consequence of the appearance of deeper channels that concentrated most of the water flow. Only rarely (Chamarcal and Sangarrén) the water sheet was homogeneously distributed in the wetlands. All reedbeds grew spontaneously after the transformation into irrigated farmland, except one (Sangarrén) which is mostly fed by groundwater. Burning is practiced every two or three years and totally or partially eliminates the plant cover. Other common practices are drying, grazing, burying or ploughing edges, patches or the entire wetland. Wetland location varied from hillsides with wastewater outcrops to flat valley floors.

Methods

Vegetation sampling

Reed structure of each wetland was studied between 20 August and 15 September 2004, during the maximum plant development period. Height, diameter, density of green stems and water depth were obtained by measuring all stems in random and side located 40 cm quadrates. Fifteen quadrates were measured in wetlands larger than three hectares, 10 in those between one and three hectares and five in those smaller than one hectare. All plant biomass was obtained by drying all aboveground plant material and detritus in the quadrate for 48 hours at 60ºC. Thirteen habitats were defined according to the abundance of *Phragmites*, *Scirpus*, *Tamarix* and ligneous species (*Suaeda* sp. and *Atriplex halimus*). Homogeneous habitat was considered when one species surpassed 90% of the habitat area. A limit of 50% of the habitat occupied by *Phragmites, Scirpus, Tamarix* or ligneous species was taken to consider one habitat as dominated by these species. Usually, the second dominant species reached more than 30% of the area of the habitat and was then considered as accompanying species and a new habitat defined. If area covered by accompanying species was lower than 30% then a new habitat of a dominant species with other species mixed was created. If a 50% of the habitat was not covered by any of the main species then a new mixed habitat was created. Plant diversity was sampled using randomly distributed transects across the wetland, three transects in wetlands <0.5 ha, five in wetlands of 0.5-1 ha and seven in those larger. The amount of transects was obtained by observing the increase in the observed variance for plant diversity with respect to the number of some preliminary transects. When the slope of the curve variance-number of transects began to decrease, that amount was selected. The landscape Shannon diversity index (H' = -$\sum p_i \cdot \log_{10} p_i$, where p_i is the area of landscape type i / total segment area) (Turner et al., 1989) was calculated by extrapolation to area from the *in situ* measurement of the length of every habitat in all transects. Bush abundance was recorded as the proportion of length occupied by bushes compared to the total length of transects.

Bird census

Absolute censuses were carried out in each wetland in the breeding and wintering periods. Counting points were distributed to cover the whole wetland area with radii of 100 meters, the largest distance from which the reed warbler (*Acrocephalus scirpaceus*; the commonest species) is easily detected. In the breeding period, they were visited three times between sunrise and the following four hours in mid April, mid May and mid June. Nine of them were visited in 2004 and all were visited in 2005 and 2006. All territorial males heard during a period of eight minutes (the first three minutes allowed the birds to acclimatize to the presence of observers) were counted and their territories were drawn on a map of the wetland in order to avoid duplicated contacts. The maximum number of territorial males of each species of bird detected on any of the three sampling dates was used in all statistical analyses. Additionally, all birds showing obvious breeding behavior (calling of chicks, nests or females with chicks) but not territorial males were counted and contrasted to the male territories drawn on the map. These territories were added if they were different from those of territorial males.

During the winter periods of 2005 and 2006 two censuses were performed. In the morning of mid January, using the same method as the breeding censuses, one visit was made to each wetland and all birds observed and heard were counted. In the evening, for one hour before and 45 min after sunset all birds entering to roost were counted. All records were made by the same observer, except those made in winter when flocks of birds arrived to roost in some wetlands and one person was unable to cover the whole surface. In these cases two additional experienced observers collaborated.

Species and community characteristics

Bird abundance was estimated as the maximum number of birds detected during the breeding season and as the total number of birds detected during both winter uses (daytime foraging and nocturnal roosting) each year averaged for the three years. Bird diversity in each wetland was estimated as the Shannon diversity index averaged for the three years. Species richness was estimated as the maximum number of species of all surveys detected in the wetland. In the analysis of the effect of burning on wetlands, bird abundance, bird diversity and species richness were estimated separately for each year. In estimating habitat requirements for species of bird, densities were calculated for the commonest species in each census (Fig. 2). The commonest species were defined as those that completed together 90% of total contacts.

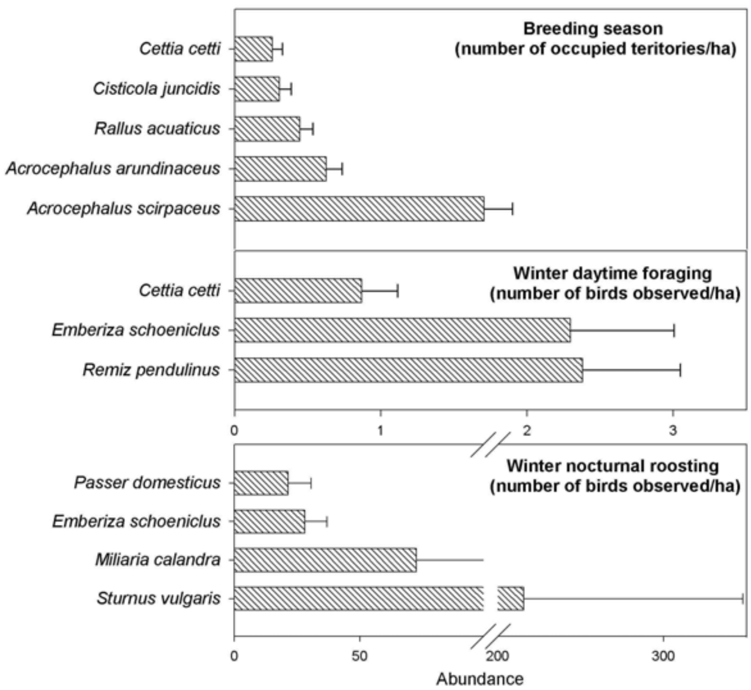

Figure 2. Seasonal distribution of the main species detected during the censuses of 15 wetlands across two study years. The species represented supposed >90% of the total observations for each census period. Bars represent average of observations for each wetland use and error bars refer to standard error (SE).

Statistical analysis

Pearson's correlation were performed with the environmental variables to eliminate those highly correlated from further analysis (p<0.01; Table 1).

After this first approach green stem diameter and wetland perimeter were definitively eliminated from the analysis because of their high collinearity with other variables. Predictive variables were related to: a) bird community variables, performing a multiple linear regression (MLR) with a backward procedure, and b) bird species densities, performing a redundancy analysis (RDA) with a forward procedure. MLR makes it possible to examine the relationships between environmental variables and each bird community variable separately. Backward procedure makes it possible to detect which environmental variables provided the greatest significance to the model. To reduce collinearity effects between selected variables, we studied the variance inflation factor and condition index for each variable of each model (Quinn and Keough, 2002). A previous detrended correspondence analysis (DCA) showed that the longest gradient was shorter than 3.0, leading us to select a linear ordination method. Within this method, constrained ordination (RDA) was selected to find the variability in species composition that can be explained by the measured environmental variables (Lepš and Šmilauer, 2003). The forward procedure and Monte Carlo permutation procedure (MCP) were implemented to detect those variables that explained the greater amount of variance. Its ordination diagram approximately shows the spatial distribution of species variables (points) among environmental variables (vectors), and the relations between species and wetland characteristics in terms of their between-site variations.

To detect any possible effect of burning on bird communities we performed a *t*-test for each sampling year and for each bird community variable. The *t*-test makes it possible to find differences between wetlands submitted to different treatments (burned/unburned) with unbalanced samples. Statistical analyses were carried out with SPSS for Windows 13.0 (SPSS Inc., 2004) and Canoco for Windows 4.5 (Biometrics-Plant Research International, 2002).

Table 1. Description and summary of the explanatory variables used for the statistical analysis.

Explanatory variable	Abbreviation	Mean	Maximum	Minimum
Habitat parameters				
Wetland area (ha)	Area	1.99	6.58	0.13
Depth (cm)	Depth	19.8	84.72	0
Perimeter length (km)	Perimeter	1.04	4.02	0.25
Compactness of the wetland patch*	Compactness	0.52	0.84	0.23
Nearest distance to a wetland > 1ha (km)	Distance 1	2.71	7.29	0.42
Nearest distance to a wetland > 3ha (km)	Distance 3	7.85	29.15	0.66
Vegetation parameters				
Height of *Phragmites* stems (cm)	Height	189	264	112
Diameter of *Phragmites* stems (mm)	Diameter	4.08	6.24	2.55
Density of *Phragmites* stems (nº/m²)	Density	112	198	71
Aboveground plant biomass (kg/m²)	Biomass	3.35	4.88	1.85
Plant diversity in vegetation transepts (Shannon diversity index, nits)	Heterogeneity	0.27	0.46	0.03
Length of area occupied by bushes in vegetation transepts (%)	Bush	5.50	46.18	0

*Measured as $2 \cdot \sqrt{\Pi \cdot A} / P$, where Π is 3.1416 (Forman, 1995)

Results

Environmental determinants

Wetlands were scattered along all environmental variables, indicating a wide covering of environmental conditions (Fig. 3). Large wetlands (Moncalver and Albalatillo) showed strong relations with the presence of ligneous species, and Sangarrén showed an important isolation from other large wetlands. Amount of bushes and height of stems were the most prominent explanatory variables for richness and diversity during the breeding season (Table 2). The most species-rich and diverse bird communities were hosted in wetlands with relatively high plant heterogeneity. Wetlands with high levels of plant biomass proved most suitable for numerous bird communities and also had a positive influence upon their richness. Long distances from other wetlands (>1 ha) had a negative influence on richness and abundance.

The winter results were less significant due probably to the lower number of censuses carried out. In this period, larger wetlands hosted the richer bird communities both during daytime foraging and nocturnal roosting (Table 2). Wetlands with a large amount of bush cover attracted more species during daytime foraging and wetlands with a high plant biomass had a similar effect during nocturnal roosting.

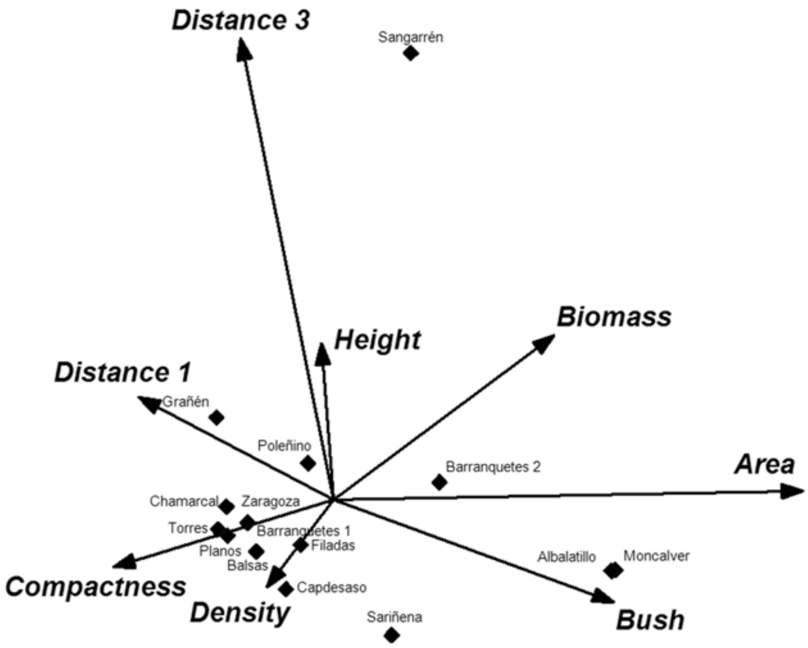

Figure 3. Principal component analysis of the environmental variables selected by the correlation analysis and redundancy analysis (RDA).

Table 2. Results of the backward stepwise multiple linear regressions explaining the relations between environmental variables and bird community variables. Only the best model obtained by the backward procedure is represented. Criteria to select variables was probability of F <0.05 and to reject F >0.1.

Use	Dependent variable	Model r²	Explanatory variable[a]	Partial correlation	Significance[b]
Breeding season	Richness	0.952	Bush	0.922	***
			Height	0.816	**
			Heterogeneity	0.733	*
			Density	0.696	*
			Biomass	0.692	*
			Distance 1	-0.661	*
	Abundance	0.523	Biomass	0.625	*
			Area	-0.532	<0.1
			Distance 1	-0.501	<0.1
	Diversity	0.865	Height	0.874	***
			Bush	0.837	***
			Heterogeneity	0.589	*
Winter foraging	Richness	0.881	Area	0.815	***
			Bush	0.587	*
	Abundance	<0.500	—	—	—
	Diversity	<0.500	—	—	—
Winter roosting	Richness	0.853	Area	0.856	***
			Biomass	0.691	**
	Abundance	<0.500	—	—	—
	Diversity	<0.500	—	—	—

[a] Definitions of abbreviations in Table 1.

[b] *p<0.05; **p<0.01; ***p<0.001

The greatest diversity of birds for all three wetland uses (breeding, winter daytime foraging and winter nocturnal roosting) were located between stems of a height of 200 and 230 cm (Fig. 4a), except for the case of the Albalatillo wetland, where plant diversity dominated the other variables (Fig. 2). Bird richness was highest for all three wetland uses between 3.5 and 4 kg/m^2 of aboveground plant biomass (Fig. 4b). In terms of area, there was constant increase in the number of species for all three uses up to the

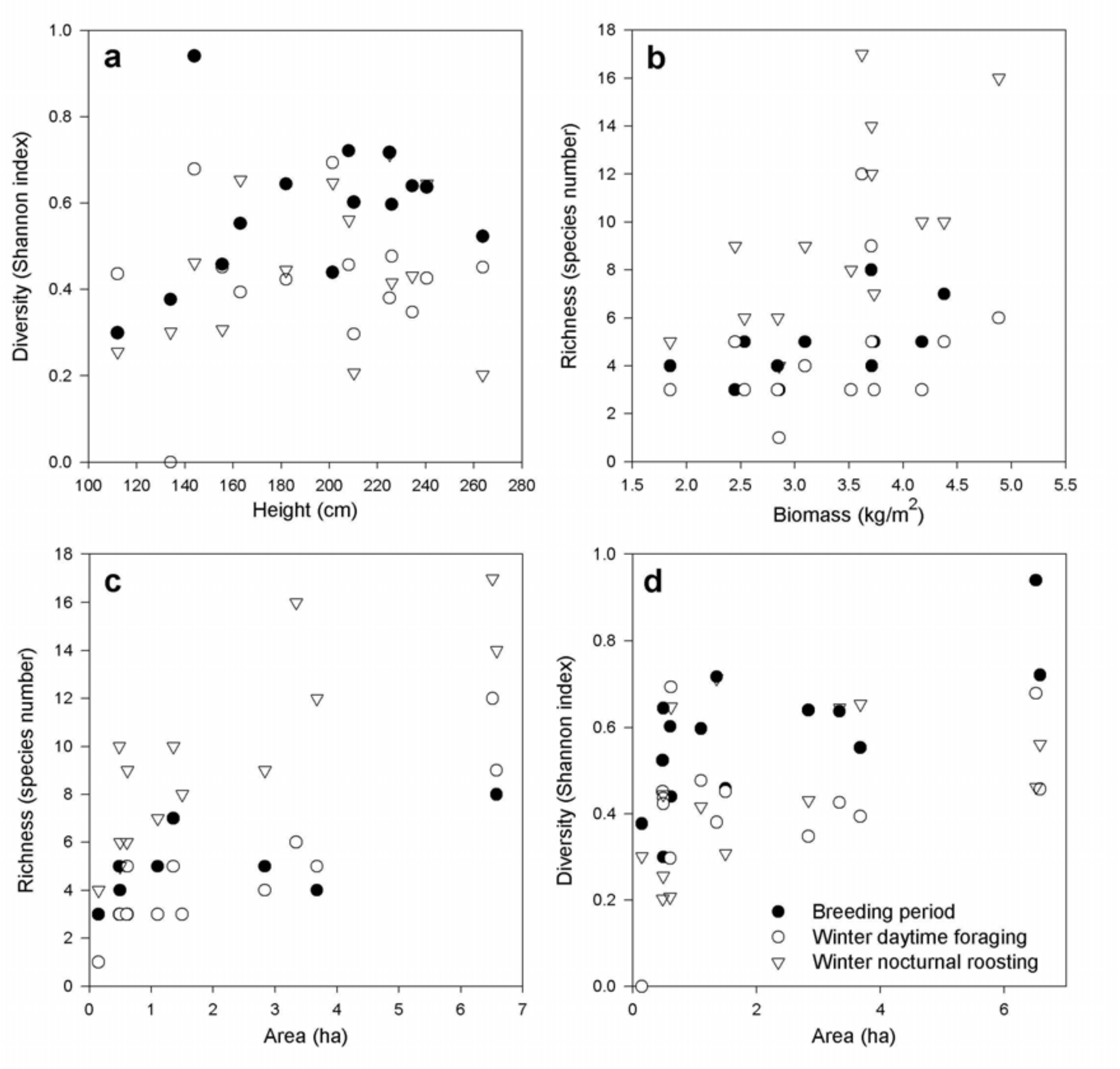

Figure 4. Relations between environmental variables and bird community variables detected as significant in the multiple regression analysis for at least one of the wetland uses. Winter censuses were carried out once per year and breeding census three times per year for each wetland.

largest wetland studied (Fig. 4c). Additionally, although not significant in the regression analysis, the graph showed that bird communities established in large wetlands were more diverse for the three wetland uses studied (Fig. 4d).

Species requirements

All the environmental variables for wetlands that proved relevant in the breeding season after the RDA were also previously selected as being significantly explanatory by the backward procedure of the MLR for some of the bird community variables. The variables selected by the forward procedure of the MCP (p<0.05) in the RDA explained 50.4% of the total variance, including 46.6% on the first two axes. Half of the wetlands were grouped around a few environmental axes while the other half were distributed along each of the environmental axes of the PCA (Fig. 2), indicating similar environmental conditions for those grouped and a wide range of environmental conditions for the remainder. RDA enabled us to interpret differences between sites in terms of species abundance (Fig. 5). The presence of bushes in wetlands favoured numerous but few abundant species that are not strictly linked to reeds [Cetti's warbler (*Cettia cetti*),

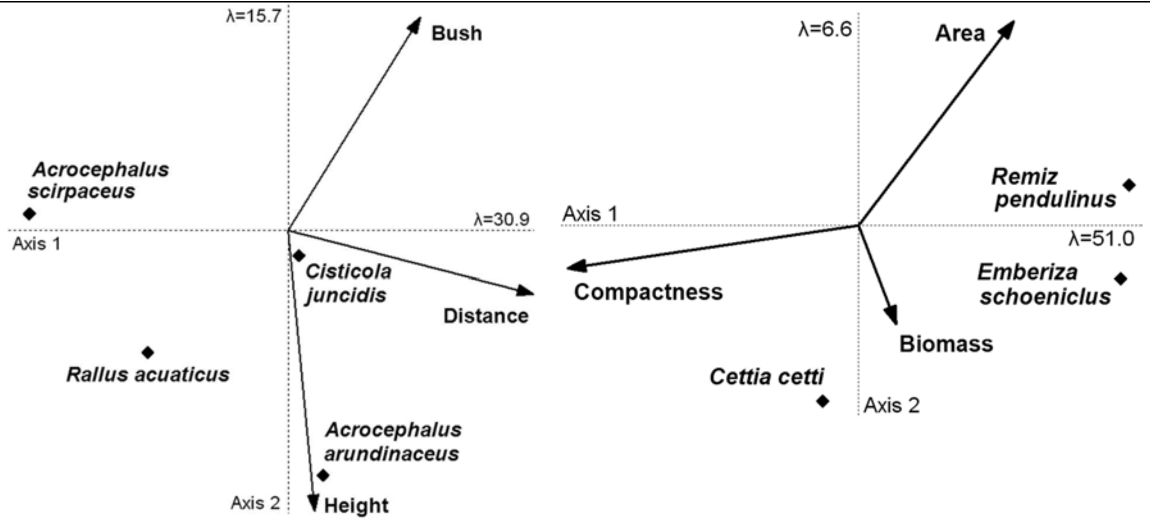

Figure 5. Ordination diagram of the commonest bird species during the breeding season (RDA). The environmental variables were those that significantly affected the bird species selected (MCP: p<0.05) (right). Ordination diagram of the commonest bird species during winter daytime foraging and winter nocturnal roosting (RDA). Environmental variables were selected with p<0.1 in the MCP to enable a better interpretation of the graphs (left). Definitions of abbreviations in Table 1.

nightingale (*Luscinia megarhyncos*) or penduline tit (*Remiz pendulinus*) as a breeder; which corresponded to <5% of birds in the census]. Homogeneous and high reedbeds favoured few but abundant species [great reed warbler (*Acrocephalus arundinaceus*) and water rail (*Rallus acuaticus*); ~10% of birds in the census for each]. In contrast, the most abundant species [reed warbler (*Acrocephalus scirpaceus*; 50% of birds in the census] was markedly influenced by the distance from large wetlands, long distances from large wetlands meant there were fewer of them. The abundance patterns of other species [e.g. fan-tailed warbler (*Cisticola juncidis*)] located in the centre of the ordination diagram could not be explained by any environmental variable.

During winter daytime foraging and nocturnal roosting few variables were considered to be significantly influential by the RDA (MCP; $p<0.05$), probably because of the lower intensity of sampling. Therefore, to make it possible to interpret the diagrams better, those with $p<0.1$ (biomass in foraging and area in roosting; Fig. 5) were also represented. The foraging diagram separated two groups of wetlands; irregular, medium or large sized ones where the most abundant and wintering species appeared regularly in these wetlands [penduline tit -50% of birds in the census - and reed bunting (*Emberiza schoeniclus*) -~40% of birds in the census -], and compact, medium or small sized ones with high aboveground biomass where less abundant and resident species appeared in this type of wetland (Cetti's warbler -8% of birds in the census). Roosting preferences did not offer much information. Some species preferred to roost in large wetlands (reed bunting) and others in smaller ones [house sparrow (*Passer domesticus*)], and other species [corn bunting (*Miliaria calandra*)] were affected by the distance from other wetlands, the longer the distance to other wetlands, the larger the roosts of this species were.

Effects of burning

Burning had severe negative effects on bird abundance leading to a drastic reduction in occupied territories ($t=3.52$; $p<0.001$; n=27) during the breeding season. Bird diversity also decreased, although less significantly ($t=1.87$; $p=0.07$; n=27), as a consequence of burning in that season (Fig. 6). No significant effects were found for either winter use (daytime foraging and nocturnal roosting) for any of the community variables.

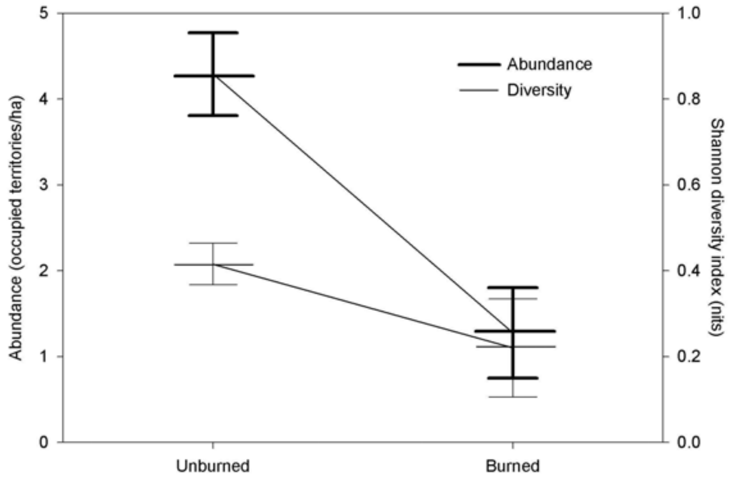

Figure 6. Variation in bird abundance and bird diversity during the breeding period in burned and unburned wetlands across three study seasons. Bars represent mean ± 1 SE. The census covered nine wetlands of ~0.1 to 6.5 ha.

Discussion

Vegetation determinants

The reedbed structure of the studied wetlands determined the richness, abundance and diversity of bird communities all year round. Similar results were found for passerine communities during the breeding season in southern and central Europe (e.g., Poulin et al., 2002; Surmacki, 2005; Baldi, 2006). Lower significance was demonstrated by the results for the winter period because of the lower number of censuses carried out. The most influential environmental variables were plant heterogeneity and the height of reeds during the breeding season and area during winter daytime foraging and nocturnal roosting. Plant heterogeneity during the breeding season was the biggest determiner of the structure of the bird community, providing a number of bush-dwelling species whose territories are in drier areas of the wetlands (mounds, shores, etc.). This situation was especially noticeable in the Albalatillo wetland where 45% of the vegetation was dominated by ligneous plants (*Suaeda* sp., *Atriplex halimus* and *Tamarix africana*), and hosted the highest richness of all sampled wetlands. Several studies have suggested the importance of the plant heterogeneity of wetlands as a factor that may have major direct effects on the bird communities of the wetlands in the reedbeds of the Ebro delta (Martinez-

Vilalta et al., 2002; Moreno et al., 2008d) and on moorlands in Scotland (Pearce-Higgins and Grant, 2006). Additionally, the number of species during winter daytime foraging was also positively affected by plant heterogeneity in a very similar way.

A height of 200 to 230 cm and an aboveground biomass of 3.5 to 4 kg/m^2 were the most suitable for bird diversity and bird abundance, respectively, during the breeding and wintering seasons. This range of biomass corresponded to an approximate density of 100 stems/m^2. This agrees with the results obtained by Ille and Hoi (1995), who highlighted an inverse relation between vegetation density and prey abundance (arthropods) in homogeneous reedbeds. This could involve a negative effect on bird diversity and abundance in areas with excessively high stem densities, which has been also noted by Báldi and Kisbenedek (1999). We hypothesized that in these situations only fitted species would easily benefit from reedbeds. In our study area these species were the reed warbler and the great reed warbler. For them, Honza et al. (1998), Foppen et al. (2000) and Batáry et al. (2004) found that stem height and density had the most important effect on breeding success, and Martínez-Vilalta et al. (2002) reported the same effect produced by stem density on the entire passerine community. The effect of biomass was similar to stem density and height allowing the use of reedbeds with high biomass by the mentioned fitted species. Bird communities of all wetlands, except one, with stems higher than 230 cm and/or with a biomass higher than 4 kg/m^2 were strongly dominated by the reed warbler and great reed warbler during the breeding season and by the reed bunting and penduline tit during winter daytime foraging, which are typical inhabitants of reedbeds. Additionally, wetlands with high aboveground biomass hosted more species during winter nocturnal roosting, probably because they offer a more compact and, therefore, safer physical structure to live in than low aboveground biomass reedbeds because these ones do not have a complex aboveground structure during winter when standing dead stems have no leaves on it. .

Habitat determinants

Isolation of wetlands (distance to wetlands >1 ha) resulted in a lower number of species and abundance of birds, which agrees with other studies of similar wetlands (Foppen et al., 2000; Paracuellos and Telleria, 2004; Paracuellos, 2006). Isolation is a consequence of the patchy distribution of wetlands and generates source-sink dynamics in most of the species of the bird communities (Pulliam and Danielson, 1991; Verboom et al., 2001). Small patches with unstable populations, located far from large wetlands and dependent on them, were less likely to be colonized by dispersal birds. Even so, more empty patches may occur simply because no species has been able to colonize them yet as a consequence of long distances (Foppen et al., 2000; Verboom et al., 2001). Larger wetlands showed lower bird abundance, which disagrees with the results reported by Paracuellos (2006) in similar wetlands in Spain. This can be explained in two different ways: first, observers did not increase their efforts sufficiently in large wetlands and underestimated bird abundance, and second, large wetlands hosted more different habitats, and some of these habitats are unsuitable for some of the species, causing a decrease in total abundance (Nee and Cotgreave, 2002). According to this statement plant heterogeneity should have influence on bird abundance which has not been detected by the MLR (Table 2). This could be due to the reduced number of wetlands with large numbers of different habitats (only Albalatillo and Moncalver) that hinder the potential effect of high plant diversity.

Bird population richness and diversity increased with wetland size whether in the breeding season or during the winter uses (e.g., Craig and Beal, 1992; Celada and Bogliani, 1993; Hawke and José, 1996; Weller, 1999; Baldi and Kisbenedek, 2000; Fig. 2). Bird diversity increased quickly along with wetland size from the smallest to ~0.6 ha. After this point, the increase fell drastically until the largest wetland. Following this trend the asymptotic maximum would probably be located at around 1 ha and minimum at around 0.5 ha, a very low value in whatever case, which is in accordance with the simplicity of these recently created wetlands. These small wetlands play a greater role in the metapopulation dynamics of bird species than their small size might imply (Gibbs, 1993), and a few studies have reported on their importance for bird communities (Celada and Bogliani, 1993; Davis et al., 2006; Paracuellos, 2006). Area also had a similar effect on richness, although the increase was smaller, in a similar way to that illustrated by Baldi et al. (2000) in Lake Velence, Hungary.

Species requirements

During the breeding season, the isolation of wetlands has a highly negative effect on the most abundant species (reed warbler) in our wetlands according to the analysis of bird communities, and this was studied in detail by Foppen et al. (2000). High amounts of bushes have a negative effect on most abundant species (great reed warbler and water rail) that typically prefer dense and tall aquatic vegetation (Perrins, 1998), which is in accordance with the observed positive influence on their abundance of the height of reed stems. This result also fits with the aforementioned situation whereby beyond the height of maximum bird diversity, only specialists could benefit from the reedbed. During winter daytime foraging, the only species that benefits from reedbeds in our study area (reed bunting and penduline tit) were less specialist inhabitants of reedbeds (Perrins, 1998). They were also the most abundant and were strongly influenced by the shape of the wetlands. The highest abundances of these species were found in more amoeboid or elongated wetlands. This could be due to the increase of the effect of edges present in these irregular shapes that favour more species than more compact wetlands as has been studied by Baldi et al. (1999; 2006) in reedbeds and by Davis (2004) in grasslands. Finally, during winter nocturnal roosting, isolation had a positive effect on the most abundant species (corn bunting), contrary to the effects found for the richness and abundance of the bird population during the breeding season.

Effects of burning

Winter burning involves severe alterations to the wetland avian community (Gabrey et al., 1999; Gabrey and Afton, 2004; Isacch et al., 2004; Brennan et al., 2005; Eades et al., 2005). In our wetlands, the drastic reduction in bird community abundance and diversity during the breeding season is a consequence of burning which is carried out in february-march, just before that season (Fig. 7). There are two particularly important and probable reasons for this reduction: first, the elimination of high amounts of insect larvae that use reed stems to survive the winter and, second, the loss of plant structures in which to install nests. The two reed warblers arrived before the reed stems were high enough to build nests. Gabrey et al. (1999; 2004) and Isacch et al. (2004) reported important changes in the migratory species that used wetlands before and after winter burning in North and South America. More research is needed to know the effects of winter burning in Mediterranean wetlands on a long-term scale.

Implications for wetland management

Based on these results some major guidelines can be proposed in order to integrate wetland management related to bird conservation into land use planning. These will contribute to manage and create suitable wetlands for the development of a bird community in accordance with the possibilities of the region. The basic idea is to achieve high plant heterogeneity during the process of designing the wetland. The first variable to define is the area of the wetland or the land area available to create the wetland. It must be always larger than ~0.6 ha. If this is not possible, it is recommendable to create compact wetlands to enable reeds to grow up to 200-230 cm with a density of ~100 stems/m^2. If a wetland larger than 0.6 ha is available, the next step will be to choose between a regular or irregular shape (lobed or amoeboid), irregular being the better option. If it must be regular, we have then to choose between a homogenous or heterogeneous vegetation, heterogeneous being the better option. Plant heterogeneity could be favoured all around the wetland creating mounds and hollows (Bruland and Richardson, 2005a). Patches of different plant species can be managed or planted in dry and flooded areas or it can be left to natural colonization. If the design allows for an irregular shape then we could choose between homogenous or heterogeneous vegetation, in the same way as for a regular shape.

Following these criteria the ideal wetland in irrigated agricultural catchments of semi-arid environments of the Mediterranean region will be a relatively large site (>0.6 ha), with a lobed or amoeboid shape and with patches of different plant communities growing within the wetland, with bush species on the shores, reeds as high as 200 m and densities up to 100 stems/m^2 and located in a mosaic of wetlands in the territory. It is also strictly necessary for successful management to avoid the uncontrolled winter burning of wetlands to allow for the natural evolution of the recently established bird communities. After future research, prescribed burnings have been considered that may be part of a long-term management strategy (Brennan et al., 2005). Many other management practices can be used efficiently to manage reed beds (Hawke and José, 1996). We considered that no burnings should be carried out before prescribed burnings were deeply demonstrated as beneficial for bird populations in these Mediterranean wetlands. Following these guidelines in semiarid zones intensively transformed for agriculture will contribute to their sustainable development through the conservation of natural resources.

Acknowledgements

This work was supported by CICYT (REN2003-03040), Aragon Government –D.G. Investigación, Innovación y Tecnología– Research Group on Ecological Restoration (E 61), Instituto de Estudios Altoaragoneses-Diputación de Huesca and Comarca de Los Monegros. We thank Boletas Birdwatching Centre (J. J. Saiz) and D. Cazo for their help during the censuses and J. M. Rey Benayas (Alcalá de Henares University) for his statistical support.

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Chapter 6

Relationships between landscape pattern, wetland characteristics, and water quality in agricultural catchments*

Abstract

Water quality in streams is dependent on landscape metrics at catchment and wetland scales. There exist relations between catchment and wetland metrics. A large pool of water quality variables (salinity, nutrients, sediments, alkalinity, other potential pollutants, and pH) has been found to be related to landscape metrics at catchment and wetlands scales in the agricultural areas of a semi-arid Mediterranean region dominated by irrigated farmlands. The percentage of arable land, landscape homogeneity (low value of Shannon index), and the number of stock farms are significantly correlated with salinity and nutrients' related variables at catchment scale. The relative abundance of wetlands and the aggregation of its patches influence salinity variables at wetland scale. The number and aggregation of wetland patches are closely correlated to the landscape complexity of catchments. These results suggest that more effective results in water quality improvement would be achieved if we acted at both catchment and wetland scales. A set of guidelines for planners and decision makers is provided for future agricultural developments or to improve existing ones.

Resumen

La calidad del agua en los cauces depende de los medidores del paisaje a escala de cuenca y humedal. Existen relaciones entre los medidores del paisaje y de los humedales. Un gran conjunto de variables de calidad del agua (salinidad, nutrientes, sedimentos, alcalinidad, otros potenciales contaminantes y pH) han resultado tener relación con los medidores del paisaje a las escalas de cuenca y humedal en zonas agrícolas de una región semiárida Mediterránea dominada por explotaciones de regadío. El porcentaje de tierras cultivadas, la homogeneidad del paisaje (bajo valor del índice de Shannon) y el número de granjas de porcino están significativamente correlacionadas con las variables asociadas a la salinidad y los nutrientes a escala de cuenca. La abundancia relativa de humedales y la agregación de sus parches influyen sobre las variables de salinidad a escala de humedal. El número y agregación de los parches de humedal están estrechamente correlacionados con la complejidad del paisaje de las

* Moreno, D., Mander, Ü., Comín, F.A., Pedrocchi, C. and Uuemaa, E. *Journal of Environmental Quality*. Submitted.

cuencas. Estos resultados sugieren que se pueden obtener mejores resultados en la mejora de la calidad del agua si se actúa tanto a escala de cuenca como de humedal. Se aporta una serie de directrices para la toma de decisiones de los gestores territoriales para futuros desarrollos agrícolas o para mejorar los existentes.

Introduction

Landscape composition and structure have an important influence on water quality in streams at catchment level (Arheimer and Liden, 2000; Gergel et al., 2002). Catchment land use has usually been highlighted as one of the most influential parameters of water quality, in particular the amount of arable land in agricultural catchments (Jones et al., 2001; Buck et al., 2004; Uuemaa et al., 2007) and the urban area in highly humanized catchments (Wear et al., 1998; Paul and Meyer, 2001; Kloiber, 2006). Agricultural and urban land uses act as sources, and riparian zones or woodlands as sinks of non-point pollution. The spatial arrangement of sources and sinks has a strong impact on water quality (Chen et al., 2002; Gergel, 2005), but when an area covered by sources exceed 65% of the catchment area, then the spatial arrangement influence decreases (Gergel, 2005). Non-point source pollution, defined as anthropogenic nutrient inputs to aquatic systems from diffuse sources within a catchment, is certainly one of the greatest problems facing water resources, and plays a central role in the water quality of rivers (Carpenter et al., 1998).

Landscape metrics can be used to quantify human impacts on streams by examining land use in surrounding areas (Gergel et al., 2002). These have commonly been used as indicators of water quality in a variety of aquatic ecosystems, such as rivers (Jones et al., 2001; Uuemaa et al., 2007), upland wetlands (Poiani et al., 1996; Houlahan and Findlay, 2004), riparian forests (Weller et al., 1998; Baker et al., 2006), estuaries (Paul et al., 2002) or lakes (Martin and Soranno, 2006). Johnson et al. (2001) have found that using only landscape measurements obtained solely from remote sensed data can explain about 75% of the water quality variability in catchments. More specifically, Jones et al. (2001) illustrated that landscape metrics consistently explained 83% of the variation of nitrogen yields to rivers, such as 73% and 79% of the variability in dissolved phosphorus and suspended sediments. Landscape indicators such as riparian buffers could be useful to control the nutrients enrichment of streams, since wide riparian zones of uniform width retain more nutrients from adjacent land uses than buffers with variable width, because some nutrient-rich fluxes can flow through gaps (Weller et al., 1998). On the other hand, buffer zones should only be established in riparian sections where the water flow is concentrated, e.g. depressions and thalwegs (Mander et al., 1997). Landscape metrics obtained using Fragstats software (McGarigal et al., 2002) are increasingly used because of their consistency in the analysis of pattern-processes relationships (Tischendorf, 2001). Nevertheless, very few studies have reported consistent relationships between the Fragstats metrics of catchments and water quality. In this sense, Uuemaa et al. (2007) reported an 84% variance of COD, explained by mean shape index and contagion metrics in agricultural catchments under boreal conditions.

Wetlands play an important role in agricultural catchments in the control and improvement of wastewater (Mitsch and Gosselink, 2000b). The capacity of wetlands to improve water quality is dependent on a large number of parameters that have been widely studied, such as vegetation cover or type, depth, water retention time, soil type, size or climatic variables (Reed et al., 1995; Kadlec and Knight, 1996; Kuusemets and Mander, 1999; Romero et al., 1999; Bachand and Horne, 2000; Moreno et al., 2007). The use of these variables, which are clearly related to certain landscape metrics, has been proposed as a better tool to study the state of water in aquatic ecosystems, but it is still poorly developed (Gergel et al., 2002). The change of scale is a common question in landscape ecology, and has been proposed to treat the problem of water quality at different scales, usually at the local and catchment scales (Gergel et al., 2002; Buck et al., 2004). In comparing scales, Gergel et al. (1999) analyzed the effect of the entire catchment and only the wetlands of the catchment upon the release of dissolved organic carbon to lakes and rivers. In addition, Buck et al. (2004) compared the effects of second and fourth order streams upon water quality. The effect of the entire catchment and of the wetlands of the catchment on a complete pool of water quality variables has not yet been reported.

The appropriate planning of wetland creation or restoration focused at the catchment scale has been proposed as a decisive tool to ensure the control and improvement of the quality of agricultural wastewater (Mander et al., 2000; Trepel and Palmeri, 2002; Zedler, 2003). In the process of planning at catchment scale, landscape metrics have a decisive influence on water quality as has been demonstrated. Could wetland features by themselves exert an influence on landscape composition and structure of catchments, however?. If that happens, how can one elucidate new complementary considerations to those obtained from the catchment and wetlands scales separately?.

The aim of this study is to ascertain the relationships between the landscape metrics of catchments, characteristics of wetlands spontaneously formed after the irrigation of agricultural fields and the water quality in streams. In order to accomplish this general aim, three steps have been implemented. The first is to select the landscape metrics of the catchment that have significant effects on water quality. The second step was to determine which wetland metrics have significant effects on water quality. To do this, the landscape metrics of wetlands were used in combination with traditional wetland metrics (e.g. depth, flow, biomass, etc.) On the basis of the gathered information, we propose alternatives to the design and creation of wetlands in semi-arid Mediterranean agricultural catchments. The third step was then to analyse how these alternatives could influence landscape metrics commonly used in landscape studies (Forman and Godron, 1986). One of the goals was to compile a set of guidelines on wetland design and creation at catchment scale, serving as a useful tool for planners and decision makers in agricultural areas.

Study area

Monegros is a 270,000 ha inland semi-arid Mediterranean region (average annual temperature 14.5ºC, average annual precipitation 400 mm) located in the centre of the Ebro River basin in NE Spain (Fig. 1), with high interannual variability (Comín and Williams, 1993; Moreno et al., 2007). Soils are mainly dominated by a Tertiary structure composed of clays with different salinization levels (conductivity ranges between 1-10 mS cm^{-1}) and inserted sandstone stratus. This agricultural landscape is composed of small plateaus with valleys in between, some gentle hills and low altitude mountain chains. Most of this land was transformed into irrigated agricultural fields between the 1950s and 1990s. Maize, alfalfa and cereals are the most common crops. Soil salinization and the abandonment of agriculture are now widespread in many parts of this region. As a consequence, the original pseudo-steppe vegetation dominated by rosemary (*Rosmarinus officinalis* L.), thyme (*Thymus* sp.) and halophytes (*Salicornia* sp.), as well as perennial grasses [*Lygeum spartium* L., *Brachypodium retusum* (Pers.) Beauv.], was transformed into a landscape dominated by halophytic species (*Tamarix Africana* Poir., *Atriplex halimus* L., *Suaeda vera* Forsskal. ex J.F. Gmelin.) in abandoned dry elevated zones and helophytes (*Phragmites australis* (Cav.) Trin. ex Steud., *Typha latifolia* L., *Scirpus holoschoenus* L., *S. maritimus* L., *Carex divisa* Huds.) in the abandoned wetter and lower zones of the valleys. In most of these zones, wetlands with permanent or intermittent surface water are formed and colonized by *Phragmites australis*, which is spreading and increasingly dominating the plant community because of its tolerance to changing water levels and salinity (Lissner et al., 1999a).

Figure 1. Location map of the study area with the main basins of the Spanish Iberian Peninsula (top). Selected subcatchments are shaded in grey. Subcatchments are included in the Flumen catchment and its drainage network (bottom).

Eighteen catchments within an 80,000 ha zone in Monegros region were selected for this study (Fig. 1). All catchments with wetlands larger than one hectare (eight in all) were included (the maximum size was ~6.5 ha).

Ten other catchments with wetland sizes between 1000 m^2 and one hectare were selected in accordance with the following criteria: similar topographic and geographic situation to catchments with large wetlands, similar catchment area, and accessibility (Fig. 2). All catchments were dominated by arable lands (77.36±13.61%), especially irrigated farms. The wetlands located in catchments received agricultural wastewater coming from irrigation surpluses through drainage channels or via groundwater and flowing into the fluvial system. The hydrological system was entirely dependent on the system of field irrigation in the surrounding area. Thus water drained into the wetlands during the meteorologically driest period. Wetland vegetation grew spontaneously after transformation into irrigated farmlands, except for one (Sangarrén), which is mostly fed by groundwater. Wetlands in this region are highly affected by human intervention. Burning is practiced every two or three years and totally or partially eliminates the plant cover. Wetland location varied from hillsides with wastewater outcrops to flat valley floors. The rapid development of livestock farms (especially pig farms) is characteristic of this area: seven of selected 18 catchments have at least one livestock farm.

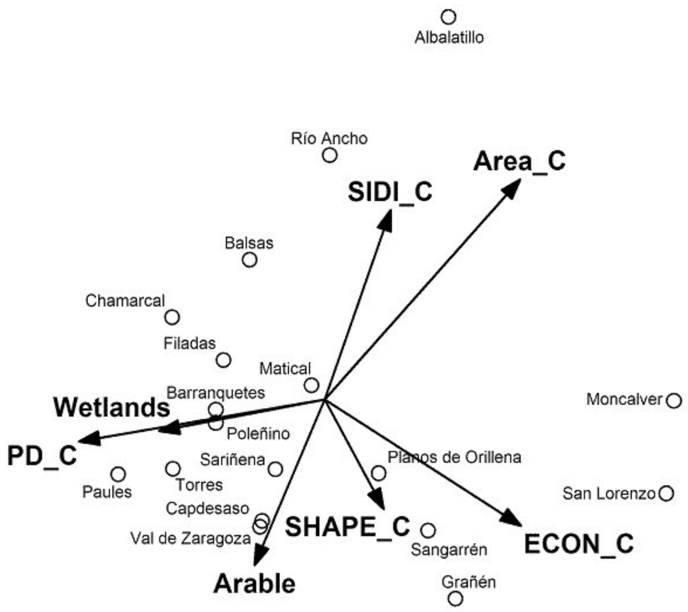

Figure 2. Principal components analysis (PCA) of the studied subcatchments using landscape metrics selected after the Pearson's correlation analysis. Definition of abbreviations are in Table 1.

Methods

Water and wetland sampling

Samples of outflowing water at the end of the catchment were collected four times per year between 2004 and 2006, once every three months. In situ field measurements of temperature, pH and electrical conductivity (EC) were performed with calibrated electronic equipment. Water samples were filtered (0.8 µm mesh size pre-combusted filters) on the same day of sampling, and total dissolved solids (TDS), total suspended solids (TSS), turbidity, alkalinity, Cl^-, Ca^{2+}, Mg^{2+}, Na^+, and K^+ were measured in the laboratory using standard methods (APHA, 1998). Nutrients (NO_3-N, Total N (TN), and Total P (TP)) were analyzed from frozen (at -30ºC) sample aliquots one month after sampling using standard methods (APHA, 1998).

Biomass, plant diversity, depth and flow of wetlands were sampled in order to ascertain possible relationships with water quality improvement. These parameters will completely join to landscape metrics the pool of wetland variables. To obtain biomass, 15 quadrates (40×40 cm) were measured in wetlands larger than three hectares, 10 in those between one and three hectares and five in those smaller than one hectare. All above-ground plant material and detritus of quadrates was dried for 48 hours at 60ºC and then weighed. Thirteen habitats were defined according to the abundance of *Phragmites*, *Scirpus*, *Tamarix* and ligneous species (*Suaeda* sp. and *Atriplex halimus* L.). Plant diversity was sampled using randomly distributed transects across the wetland, three transects in wetlands <0.5 ha, five in wetlands of 0.5-1 ha and seven in larger wetlands. The number of transects was obtained by observing the increase in the observed variance for plant diversity with respect to the number of certain preliminary transects. When the slope of the curve variance-number of transects began to decrease, that amount was selected. The Shannon diversity index (H' = - $\sum p_i \cdot \log_{10} p_i$, where p_i is the area of landscape type *i*/total segment area) (Turner et al., 1989) was calculated using the *in situ* measurement of the length of each habitat in all transects, and the value was extrapolated to the entire area. The depth of wetlands was taken at the deepest points during each water sampling and averaged for all of them. To estimate the water flow crossing the wetland, a salt solution (NaCl) of known concentration and volume was added to the water flowing in the outlet of the wetland and later using the principle of salt conservation (Comín et al., 2001).

Land cover and landscape metrics

Eleven land uses (irrigated farmlands, rain-fed farmlands, fallow lands, abandoned lands, livestock farms, arboreal vegetation, dry shrubland-grasslands, erosion deserts, wetlands, irrigation channels, and urban areas) were digitalized in all 18 catchments using aerial photographs with 0.5 m resolution taken in 2003 and ArcGIS 8.3 (ESRI, 2002). The land cover types in the photographs were checked in situ by direct observation and their areas estimated. From this cartography the ratios of arable land, natural vegetation, and area of stock farms and wetlands were obtained in order to ascertain relationships between land use and water quality at catchment level.

Landscape metrics were obtained using the computer programme Fragstats (McGarigal et al., 2002) after rasterizing land use cartography. A large pool of landscape metrics was compiled to cover the wide variation that occurred in catchments and wetlands features, which could potentially affect water quality variables (Table 1). Most of the metrics selected have typically been used in landscape analysis (e.g. Hargis et al., 1998; Li et al., 2001; Tischendorf, 2001; Gergel et al., 2002; Uuemaa et al., 2007). With this selection of metrics, landscape composition and spatial configuration were widely represented. The first uses the proportional abundances (ratios) of those classes that presumably have the greatest influence on water quality (arable land, natural vegetation, and wetlands) and a diversity index. The second determines a complete patch characterization (number, density, edges, and shape), the degree of isolation, differences among patches (contrast) and the spatial distribution of the patches (contagion) (McGarigal et al., 2002). In the study at wetland level, only metrics related with wetland designing were selected. We considered these to be priority features for the designation of the number, spatial distribution, size and shape of wetlands. Differences between wetlands represented the natural variability under the study area conditions specially affected by irrigation regime (Table 2).

Table 1. Description of used variables at catchment and wetland scale. Parenthesis in the first column indicates abbreviations of variables and parenthesis in the first paragraph of description column indicates units.

Metric	Description
Catchment	
Catchment area (Area_C)	Total area of the catchment (ha).
Wetland ratio (Wetlands)	Ratio of wetlands area in the catchment (%).
Natural vegetation ratio (Vegetation)	Ratio of natural vegetation area in the catchment (%).
Arable land (Arable)	Ratio of arable land (rainfed and irrigated) in the catchment (%).
Stock farm area (Farm)	Total area of occupied by stock farms in the catchment (ha).
Number of patches (NP_C)	Total number of patches in the landscape (unitless).
Mean patch area (PS_C)	Mean patch area for the landscape (ha).
Edge density (ED_C)	Total length of all edge segments per ha for the landscape ($m \cdot ha^{-1}$)
Patch density (PD_C)	Number of patches per unit area ($number \cdot (100\ ha)^{-1}$).
Mean shape index (SHAPE_C)	A patch-level shape index averaged aver all patches in the landscape (unitless): $$\sum_{i=1}^{m}\sum_{j=1}^{n}\left(p_{ij} \big/ 2\sqrt{\pi \cdot a_{ij}}\right) \Big/ N$$ where p_{ij} is the perimeter and a_{ij} is the area of the patch, and N is the total number of patches in the landscape.
Contagion (CONTAG_C)	Indicates the aggregation of patches (%): $$\left[1+\sum_{i=1}^{m}\sum_{k=1}^{m}\left[(P_i)\left(g_{ik} \Big/ \sum_{k=i}^{m} g_{ik}\right)\right] \cdot \left[\ln(P_i)\left(g_{ik} \Big/ \sum_{k=i}^{m} g_{ik}\right)\right] / 2\ln(m)\right](100\%)$$ where P_i is the proportion of the landscape occupied by patch type i; g_{ik} is the number of adjacencies between pixels of patch types (classes) I and k based on the *double-count* method; and m is the number of patch types in the landscape.
Euclidean nearest neighbour distribution (ENN_C)	A patch-level distance (m) to the nearest neighbouring patch of the same type, based on the shorter edge-to-edge distance is averaged over all patches: $$\sum_{i=1}^{m}\sum_{j=1}^{n} h_{ij} \big/ N$$ where h_{ij} is distance from the patch ij to nearest neighbouring patch of the same type (class), compute from cell centre to cell centre, and N is the total number of patches in the landscape.
Edge contrast index distribution (ECON_C)	Measures the degree of contrast between each patch and its immediate neighbourhood for all patches in the landscape (%): $$\left[\sum_{i=1}^{m}\sum_{j=1}^{n}\left[\sum_{k=i}^{m}\left(p_{ijk} \cdot d_{ik}\right) \Big/ p_{ij}\right] \Big/ N\right](100\%)$$ where p_{ijk} is the length (m) of edge of patch ij adjacent to patch type (class) k, d_{ik} is the dissimilarity (edge contrast weight) between patch types i and k and p_{ij} is the length (m) of perimeter of patch ij.
Simpson diversity index (SIDI_C)	Indicates the landscape composition at the landscape level (unitless): $$1-\sum_{i=1}^{m} P_i^2$$ where P_i^2 is the proportion of the landscape occupied by patch type (class) i.
Wetlands	
Wetland proportion (Area_W)	Ratio of wetlands area in the catchment (%).
Wetland area (CA_W)	Total area of wetlands in the catchment (%).
Number of patches (NP_W)	Total number of patches identified as wetlands in the landscape (unitless).
Patch density (PD_W)	Number of patches identified as wetlands per unit area ($units \cdot (100\ ha)^{-1}$).
Edge density (ED_W)	Length of all edge segments per ha of patches identified as wetlands ($m \cdot ha^{-1}$)
Mean patch area (PS_W)	Mean patch area of patches identified as wetlands (ha).
Aggregation index	Provides the frequency with which different pairs of patch identified as wetlands

(AI_W)	appear side-by-side on the map (%): $\left[g_{ii} / \max - g_{ii}\right](100\%)$ where g_{ii} is the number of like adjacencies (joins) between pixels of patch type (class) i based on the *single-count* method and max-g_{ii} is the maximum number of like adjacencies (joins) between pixels of patch type (class) i based on the *single-count* method.
Perimeter-area ratio (PARA_W)	Measures the shape complexity of patches identified as wetlands (unitless): $\sum_{j=1}^{n}\left(p_{ij} / a_{ij}\right) \Big/ n_i$ where p_{ij} is the perimeter (m) of patch *ij* (*i* type as wetlands), a_{ij} is the area (m^2) of patch *ij* and n_i is the number of patches of type *i* (wetlands).
Shape index (SHAPE_W)	Is another measure of shape complexity that reduces the variation of the index with patch size as happens with perimeter-area ratio (unitless): $\sum_{j=1}^{n}\left(p_{ij} / 2\sqrt{\pi \cdot a_{ij}}\right) \Big/ n_i$ where p_{ij} is the perimeter and a_{ij} is the area of the patch, and n_i is the total number of patches identified as wetlands.

Statistical analysis

After normality tests, only Ca, TSS, and stock farms could not be included in the main analysis. A Spearman's correlation analysis was performed between them, and between stock farms and TP. With all the rest of the variables, a previous Pearson's correlation analysis was conducted in each of the three pools of variables (water, catchment and wetlands) in order to remove those variables that were highly correlated (p<0.01), and reduce the possible effects of colineality in further analysis. To determine the variability of the cases represented by catchments, a principal component analysis (PCA) was performed with the selected variables after the correlation analysis. Multiple linear regressions (MLR) according to a stepwise backward procedure were used separately for each dependent variable at the three statistical steps of the study. The first step considered the relationship between water variables (dependent) and catchment features (predictive), while the second considered the relationship between water variables (dependent) and wetland features (predictive). The third related the most relevant landscape metrics from catchments (dependent) and wetlands (predictive) selected from the two previous MLR. This analysis could inform us about which wetland metrics could potentially influence catchment metrics related to water quality improvement. The backward procedure makes it possible to detect which predictive variables were of greatest significance to the model. Finally, another PCA was performed between catchment and wetland variables to determine further possible relations. All analyses were carried out using SPSS 13.0 (SPSS, 2004).

Table 2. Descriptive variables of wetlands located in each catchment. Numbers of landscape variables are results as obtained in Fragstats, in the rest of variables are the mean of all samples obtained.

Catchment	Area_W (ha)	NP_W (number)	SHAPE_W (unitless)	PARA_W (unitless)	AI_W (%)	PD_W (units·(100 ha)$^{-1}$)	ED_W (m·ha^{-1})	PS_W (ha)	Biomass (kg·m^{-2})	H' (unitless)	Depth (cm)	Flow (l·s^{-1})
Planos	0.46	1	1.24	729.17	99.28	0.9211	3.095	0.4608	2.84	1.06	0	0.37
Af. Val de Zgz	1.60	2	1.70	841.27	97.90	1.7715	10.3455	0.7986	2.53	1.51	6	0.86
Sangarrén	3.34	1	2.50	549.29	98.33	0.6783	12.4258	3.3352	4.88	0.27	23	1.09
Albalatillo	7.31	6	2.09	1438.28	96.61	1.1679	11.6946	1.2188	3.62	1.42	8	3.34
Moncalver	18.88	5	3.02	810.36	96.18	0.1965	6.3202	3.7758	3.71	1.04	85	6.07
Río Ancho	2.32	1	2.59	681.94	97.88	0.5885	9.3216	2.3228	2.64	0.58	5	3.02
Af. Filadas	1.19	1	4.05	1493.79	94.27	0.4575	6.6422	1.5966	4.17	0.33	32	0.87
Matical	3.19	2	2.81	877.85	96.54	0.3106	2.1496	0.6755	3.27	0.35	9	28.39
Paúles	2.03	3	2.06	1386.51	96.24	0.8925	3.32	0.4848	2.23	0.25	3	6.81
San Lorenzo	0.48	1	1.33	767.33	99.02	6.8778	77.8563	0.8125	1.85	0.31	2	0.06
Chamarcal	3.25	4	2.70	1626.77	94.08	2.4773	19.6204	0.4476	2.45	0.22	25	3.76
Barranquetes	0.90	2	2.90	2211.55	93.13	8.945	39.1792	1.1664	2.85	0.68	45	1.52
Balsas	2.33	2	1.21	10186.07	99.43	1.8249	32.4095	4.1888	1.85	1.00	10	2.11
Capdesaso	4.19	1	2.17	423.99	98.85	7.0151	45.598	0.8616	3.71	0.92	4	2.40
Poleñino	1.72	2	1.85	1255.74	97.73	0.4609	4.1298	0.8754	3.73	0.67	7	3.66
Sariñena	1.75	2	2.50	1166.87	96.35	4.5497	53.8681	1.4512	4.38	0.92	56	0.08
Torres	1.45	1	2.45	815.88	97.55	3.3828	46.8851	1.9264	3.52	0.84	23	1.74
Grañén	3.85	2	2.54	1213.90	97.40	4.4341	78.9273	1.1916	3.09	0.59	13	0.40

Results

Catchment level

Landscape metrics showed a significant effect upon water quality variables. Landscape diversity and arable lands exerted a high positive influence upon all salinity-related variables [Na, EC (r^2=0.792), and Cl (r^2=0.546); Table 3]. MLR also showed a less significant and negative effect than catchment shape on all these variables. Patch size had a significant positive effect upon nitrogen concentration-variables (Table 3). TN showed an almost complete linear dependence with NO_3 (r^2>0.99). Arable lands were strongly and negatively correlated with area covered by natural vegetation (p<0.0001). Also, patch density had a positive effect on NO_3. Spearman's correlation showed the positive significant effect (p<0.01) of livestock farms area upon TP in water. Of the rest of water quality variables, most of those that showed a strong correlation with Na provided highly similar results for the MLR with the same dependent variables [TDS (r^2=0.864), SO_4 (r^2=0.774), and Mg (r^2=0.733)]. The value of pH yielded significant positive relations with edge contrast and landscape diversity and negative relations with patch size. The variability of catchments in the study area was correctly represented (Fig. 2): no significant grouping of variables appeared, and the two first axes explained 86.4% of the total variance.

Table 3. Results of the backward stepwise multiple linear regressions explaining the relations between water quality variables and landscape metrics. Only the best model obtained by the backward procedure is represented. Criteria to select variables was probability of F <0.05 and to reject F >0.1.

Water variable	Model r²	Adjust r²	Explanatory Variable [a]	Partial correlation	Significance [b]
Na^+	0.718	0.631	SIDI_C	2.647	***
			Arable	2.249	**
			SHAPE_C	-.302	<0.1
			PD_C	-.266	0.115
NO_3-N	0.488	0.378	PS_C	1.130	**
			PD_C	0.654	<0.1
			ECON_C	-0.297	0.185
pH	0.585	0.497	ECON_C	0.681	**
			PS_C	-0.459	*
			SIDI_C	0.408	*

[a] Definitions of abbreviations in Table 1.

[b] *p<0.05; **p<0.01; ***p<0.001

Wetland level

Wetlands features showed a significant influence on some water quality variables (Table 4), especially those related to salinity [Na, EC (r^2=0.560), and Cl (r^2=0.612)]. Patch size and the number of wetland patches had a positive significant effect on Na and flow, since the wetland ratio had a negative influence on these parameters. Very similarly to Na, K concentration was positively affected by wetland ratio and negatively by patch size, but also negatively by the depth of wetlands and, less significantly and positively, by plant diversity. Notably, no effects were found on nutrient concentrations. Of the rest of variables that occurred at catchment scale, most of those highly correlated with Na were also significantly affected by virtually the same wetland features and the same sign [TDS (r^2=0.667), SO_4 (r^2=0.747), and Mg (r^2=0.423)]. In combination with the variables correlated with Na, aggregation index had a positive and significant effect on TDS. The pH value showed a very high dependence on wetland features, demonstrating a positive relation with plant diversity, wetland ratio and depth, and a negative relation with biomass and the aggregation of wetland patches.

Table 4. Results of the backward stepwise multiple linear regressions explaining the relations between water quality variables and wetland features. Only the best model obtained by the backward procedure is represented. Criteria to select variables was probability of F <0.05 and to reject F >0.1.

Water variable	Model r²	Adjust r^2	Explanatory Variable [a]	Partial correlation	Significance [b]
Na^+	0.586	0.414	PS_W	0.525	*
			Flow	-0.495	*
			NP_W	0.480	*
			Area_W	-0.355	<0.1
			AI_W	0.309	0.176
K^+	0.592	0.466	PS_W	0.587	**
			Depth	-0.530	*
			Area_W	-0.438	*
			H’	0.358	<0.1
pH	0.903	0.863	Biomass	0.709	***
			H’	-0.626	***
			Area_W	-0.547	***
			Depth	-0.491	***
			AI_W	0.438	**

[a] Definitions of abbreviations in Table 1.

[b] *p<0.05; **p<0.01; ***p<0.001

Catchment and wetland relationships

Wetland metrics exerted a significant influence upon catchment metrics (Table 5). The patch density of the catchment was positively and significantly affected by wetlands' ratio and significantly negatively affected by wetland patch size. Perimeter-area ratio and the aggregation of wetland patches also had a negative and significant effect on the patch density of the catchment. The patch size of the catchment showed a positive and significant dependence on the number of patches, and a negative dependence on wetland area. A less significant influence was produced by the patch size of the wetlands. The contrast between patches of catchments had a single, negative, and significant effect on wetland area. The patch diversity of catchments was affected by a non-linear but significant dependence (cubic) exerted by the number of wetland patches. The PCA highlighted the very strong effect exerted by wetland area and number of patches upon most catchment metrics (Fig.3). The size of wetland patches and their aggregation also produced important effects on them. These results fitted well with the results obtained from the MLR analysis.

Table 5. Results of the backward stepwise multiple linear regressions explaining the relations between landscape metrics of catchments and wetlands. Only the best model obtained by the backward procedure is represented. Criteria to select variables was probability of F <0.05 and to reject F >0.1. For SIDI_C, best curvilinear regression model is represented.

Metric	Model type	Model r^2	Adjust r^2	Explanatory Variable [a]	Partial correlation	Significance [b]
PD_C	Linear	0.692	0.597	Area_W	0.868	***
				PS_W	-0.674	**
				PARA_W	-0.509	*
				AI_W	-0.481	*
PS_C	Linear	0.532	0.388	NP_W	0.479	*
				Area_W	-0.463	*
				PS_W	0.372	<0.1
				AI_W	0.343	0.136
ECON_C	Linear	0.501	0.470	Area_W	-0.708	***
SIDI_C	Cubic	0.520		NP_W		*

[a] Definitions of abbreviations in Table 1.

[b] *p<0.05; **p<0.01; ***p<0.001

Discussion

Water quality at catchment scale

The effect of upstream land uses on water quality is well known in agricultural catchments, and has been highlighted as one of the most relevant metrics (Jones et al., 2001; Gergel et al., 2002; Uuemaa et al., 2007). This is a consequence of the higher effects of dragging produced by the rainfall on the surface materials of the bare soils of arable lands. This effect was especially severe in our study area because of the high concentration of salts in the soil matrix and its low water permeability. Salinity and related variables (EC and Na and Cl concentrations) decreased the water quality. Landscape heterogeneity also had an important effect on water salinity (Table 3). This was due to the increasing Simpson diversity index found in catchments with largest surfaces covered by irrigated and rain-fed fields. In these cases, two land-use types occupied ca 80% of the total catchment area, thereby increasing the index. This situation occurred in large catchments where areas of rain-fed fields remained untransformed by irrigation. The lower negative effect of catchment shape on saline-related variables could be explained as the consequence of the shorter distance of drainage water to the main stream in catchments with less compact shapes (high values of shape index): it reduces the possibility of water being charged by available soil solutes. In large catchments (>100 ha) where the salinity problem was especially relevant, any measure to control it should be implemented at smaller catchment sizes with lower proportions of arable lands. Due to the strong negative correlation between arable lands and natural vegetation areas, the increase of patches covered by natural vegetation of any type will also contribute to the reduction of water salinity.

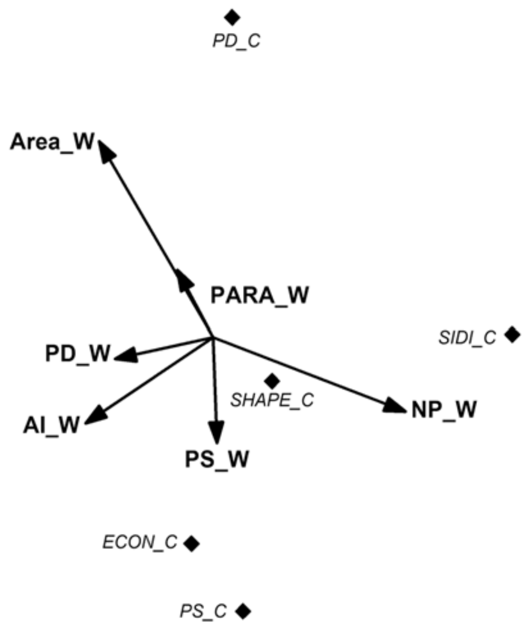

Figure 3. Ordination diagram of the selected landscape metrics of the catchments (PCA). The wetlands variables were those selected by the Pearson’s correlation analysis and included in the two first axes (86.5% of variance explained). Definitions of abbreviations in Table 1.

Patch size explained most of the variation of nitrogen concentration in streams. Large patches provided more nitrate and TN to water. These results fit with those illustrated by Chen et al. (2002) and Gergel (2002), which suggested that higher nutrient releases are found in homogenous landscapes. High mean patch size is a clear indicator of landscape homogeneity. Usually these large patches were covered by arable lands. This also fits the results that usually relate nitrogen release to the ration of agricultural land in rural catchments (e.g. Arheimer and Liden, 2000; Mander et al., 2000; Jones et al., 2001; Poor and McDonnell, 2007). Patch density also had a positive effect on nitrogen concentration. This result is contrary to those found in similar studies (Uuemaa et al., 2005). The smallest catchments (<60 ha) hosted the largest patch density values, and in cases where there were naturally few patches in a small area, the addition of one new patch supposed an important increase in the metric. This could be attributed to the effect of scale; the small catchments would be excessively small compared to larger ones to be included at the same scale. In these small catchments the distance nitrate had to traverse to reach the stream was very short, and the amount of wetlands to potentially retain water was also very small. Thus nitrate more easily reached the main stream, and its concentration was comparatively higher than in larger catchments. The presence of farms in the catchments had a definitive effect on the TP concentration of the water. This very strong relation probably hindered finding other potential relations with catchment metrics. This result highlighted how potential point-pollution could dramatically spoil water quality and mask other relevant processes. This situation has often been reported for TP in agricultural catchments with other point-pollution situations arising from human activities (Paul and Meyer, 2001; Griffith et al., 2002; Uuemaa et al., 2007).

The important effect exerted by edge contrast on water pH could be explained as a consequence of the longer distance water must cover to avoid strong edges between patches with high contrast. Along this longer trip, alkalinity increases progressively in the mass of running water, increasing pH. This increase in pH could have important effects on nitrogen removal. When this water reaches quieter places such as wetlands, denitrification could be enhanced by the high values of pH and nitrate in the water (Gale et al., 1993; Davidsson and Stahl, 2000). All three water quality variables highly correlated with Na (TDS, SO_4 and Mg) showed the highest dependence on landscape diversity (SIDI_C) and arable lands. These two variables had a strong effect on the overall quality of water in the streams. This reinforces the recommendation of reducing the size of large catchments to implement measures to improve water quality. On the other hand, smaller wetlands can loose their ability to buffer polluted water sooner than larger ones (Arheimer and Wittgren, 2002).

Water quality at wetland scale

A similar pattern of increasing water salinity variables (Na, Cl, EC, and K) with patch size and the number of patches was repeated at wetland scale (Table 4). The cause was similar to that at catchment scale; high numbers of large patches covered by wetlands were in large catchments with high water salinity. But in this case there was an important difference: a negative effect of wetlands ratio was found. This means that the larger the ratio, the lower the salinity of water. This pattern is significant but could possibly be stronger if the sampling effort were increased, for example by increasing the number of sampled catchments. The negative effect of flow on salinity could be explained as an effect of "dilution" produced by the abundant watering that occurred upstream in some of the catchments with higher flows. In these catchments, water easily passes through the sandy-gravel layer with low salt concentration and reaches the impervious layer of lutites (sodium clays) seeping quickly into the stream. "Dilution" effects have been associated to seasonal changes as the consequence of storm events (Poor and McDonnell, 2007), but not to watering regimes.

It is important to point out the absence of significant relationships between wetland metrics and nutrient-related variables, especially nitrogen variables. These results could be due to two main reasons, a) the reduced number of sampled catchments, and b) the current existing wetlands are not functional enough in nutrient retention, as a consequence of its lack of design with this purpose. In relation to the second reason, current wetlands appeared spontaneously as a consequence of transformations to irrigated farms, and usually they do not have large surfaces of water more or less homogeneously distributed in a well-structured plant community. Main streams, deeper and steeper, have appeared progressively within the wetlands, reducing retention times and subsequently the retention of nutrients. Well-designed wetlands should be created in these catchments to fulfil this crucial purpose (Moreno et al., 2007). Montreuil et al. (2006) found an inverse relation between the relative area of wetlands and the concentration of nitrate.

The three water quality variables highly correlated with Na (TDS, SO_4, and Mg) showed the highest dependence on the size of wetland patches (positively) and wetlands ratio (negatively). The positive relationship between wetland patch size and Na-correlated variables, similar to that which happened at catchment scale, do not give us the relevant information to propose measures to improve water quality. In these catchments the recommendation is, anyway, to increase the current wetland ratio (2.83±2.71%). The high values of TDS associated to those catchments that contained wetlands with higher aggregation index show us that scattered wetlands reduce more TDS than those that are more compact. TDS was mainly composed of $CaCO_3$, NaCl, $MgCl_2$, and sulphates in the water of our study areas, and consequently being an important indicator of water quality. The effect of wetlands on TDS reduction has not been comprehensively studied. Borin et al. (2001) demonstrated that losses of TDS were lower in drainage lysimeters with experimental treatment than those with crop treatments. The pH had important relations to wetland metrics. Wetlands with high biomass, low plant diversity, deep and high aggregation had a slightly but significantly higher (~0.3 more) pH value. These wetlands mostly corresponded in our study area to one sort of wetlands. They had compact shapes covered by a homogenous reedbed of *Phragmites australis* and a flat bottom that allows the accumulation of a more or less extensive water sheet. In these wetlands the decomposition of leaves and roots releases cations (Na and Ca) into the water. As the water residence time is longer, water has more time to interact with these materials, increasing its pH (Menendez et al., 2001). Additionally, soils also provide cations to accumulated water, reinforcing the effect. Most of these wetlands were also small, which also fits with the negative relationship discovered between pH and the proportion of wetlands. This situation favours denitrification processes, as explained previously at catchment scale. Consequently, promoting this soft increase of pH by creating densely vegetated wetlands with a flat bottom facilitating the accumulation of water, we were able to enhance the denitrification of agricultural wastewater.

Relationships between wetland and catchment metrics

Wetland metrics showed important influences on catchment metrics directly related to water quality. The ratio of the area of the catchment covered by wetlands and the size and number of wetland patches proved to be the most influential metrics upon most of the landscape metrics of the catchments. The positive effects on patch density and negative on patch size illustrate the direct influence of wetland area on the complexity of the landscape of the entire catchment (Table 5; Fig. 3). This result supports the idea of getting more complex landscapes to improve water quality (Chen et al., 2002; Gergel et al., 2002). However, wetland ratio also had a soft and negative effect on the shape of catchments, which means that more compact catchments hosted more wetlands in proportion to their sizes. This informs us of the higher facility that had medium-small and compact catchments to host spontaneously arisen wetlands. Most of these catchments had a similar geomorphologic structure: A 2-3 m deep sandy-gravel permeable layer above an impervious layer on slopes forms a seeping area when permeable layer crops out on the surface. In most catchment areas it occurs more or less in the middle of the area. It indicates to us the importance of these seeping areas, and advises us to use them to create new wetlands or re-design existing ones. This proposal is in accordance with that illustrated by Blackwell et al. (1999), where footslope seepages are considered as efficient buffer zones to remove nitrate in agricultural areas. Mitsch et al. (2000b) argued that wetlands in steep areas were quite susceptible to activities upstream and less likely to provide ecosystem functions, but they considered them to be valuable to control the high erosion rates that pollute water downstream with suspended solids and leached soils chemicals. The strong effect of wetlands’ ratio on edge contrast was due to the great difference that existed between wetlands and the matrix of the landscape, which is mainly composed of irrigated and rain-fed farmlands. A possible alternative to reduce this abrupt contrast would be the creation of belts of simple natural vegetation (grasslands or shrublands), having a lower contrast with the farmlands surrounding the wetlands (Lowrance and Sheridan, 2005).

Contrary to what happened with wetland ratio, the mean patch size of wetlands had a negative effect on landscape complexity (Table 5; Fig. 3). Large wetland patches reduce the density and increase the mean size of the patches of the catchments. This happened essentially in small catchments (<60 ha) which could be affected by a scale effect in regard to the largest ones, as mentioned at catchment scale. Considering landscape complexity valuable for water quality improvement, the mean patch size of these small wetlands should be smaller. The number of patches of wetlands had an effect on the mean patch size of the entire catchment. This happened especially in larger catchments with high patch sizes and where wetlands were more numerous. The diversity of the catchment was also positively influenced by the number of patches. As heterogeneity has previously been considered as important for water quality, many wetland patches are more advisable than few. In this sense, although less significantly, the negative relation between aggregation index and patch density of the catchments can be pointed out. More aggregated patches of wetlands reduce patch density. Thus more disaggregated and numerous patches covered by wetlands enhance those metrics of the catchments related with water quality (Uuemaa et al., 2007).

Conclusions and recommendations

Landscape metrics at catchment and wetland scale have a significant influence on water quality in streams. This is based on important relations between wetland and catchment metrics that could influence water quality in streams. At catchment scale, reducing the area of arable land can improve the water quality of streams. This can be fulfilled in two ways, a) by reducing the sizes of catchments dominated by arable lands (with a tentative threshold of 100 ha in our study area) and/or b) increasing the area covered by natural vegetation (grasslands, shrublands, woodlands, or wetlands). A reduction of the landscape homogeneity (directly related with water quality), in catchments highly dominated by arable lands, will also be achieved by promoting patches of natural vegetation. Additionally, specific measures should be taken to control point pollution produced by stock farms.

More wetlands should be created in agricultural catchments than currently exist (2.83±2.71% of the total catchment area). These new wetlands should be well-designed to remove nutrients. Of all guidelines known to enhance nutrient retention, they are proposed wetlands with dense vegetation and flat bottoms to enhance denitrification. Also, dilution associated with water regimes of larger wetlands which retain large amounts of water, can play an important role. Scattered and numerous wetlands are better than few and aggregated ones because within the whole catchment they will increase landscape complexity (patch density and heterogeneity) and accordingly reduce the amount of TDS in water. Groundwater seeping areas in catchments appeared to be relevant places to create new wetlands or re-design existing ones. Finally, the creation of a belt of natural vegetation (grassland or shrubland) surrounding wetlands will reduce the excessive contrast between wetlands and the matrix of agricultural farmlands.

This set of guidelines on wetland design and creation at catchment scale could serve as a useful tool for planners and decision makers in agricultural areas. This will be especially relevant in new transformations to irrigated farmlands or in the restoration of existing ones, not only by improving water quality but also by creating new landscape integrated habitats in the degraded areas of a semi-arid Mediterranean region.

Acknowledgements

This work was supported by CICYT (REN2003-03040), Aragon Government –D.G. Investigación, Innovación y Tecnología– Research Group on Ecological Restoration (E 61), DGA (CONAID)–CAI (CA1/07), Estonian Science Foundation grant No. 6083, and Target Funding Project No. 0182534s03 of the Ministry of Education and Science, Estonia.

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Chapter 7

A simple suitability model for constructed wetland siting in catchments degraded by intensive agricultural use*

Abstract

Simple tools and accessible information are needed by environmental planners to locate sites for the construction or restoration of surface flow wetlands (SFW). A flexible suitability model for SFW construction is demonstrated in small (20-2000 ha) agricultural catchments in the Ebro basin (NE Spain). The model used improved existing data layers (soil and geomorphology), simple geographical transformations (slope and distance to permanent streams) and other created data layers (land use). Detailed scales of data layers (~1:5000 and <30 m cell-size) are needed to work with small catchments. A thorough knowledge of the study area is a requirement for reducing the subjectivity associated with experts' decision. The studied cases proved that 38% of catchment areas were suitable for wetland construction activities, and another 15% were very suitable. Large catchments (>150 ha) hosted half as much suitable area as medium (60-150 ha) and small (<60 ha) catchments. Most of the suitable area was concentrated in the lower parts of the catchments examined in the study. There is enough very suitable area in all catchments to fulfil the requirements of SFW construction to improve water quality. The model is a simple and useful tool for environmental planning in areas degraded by intensive agricultural use.

Resumen

En planificación ambiental son necesarias herramientas simples e información accesible para localizar lugares adecuados para la construcción o restauración de humedales de flujo superficial. En este estudio se propone un modelo flexible para la construcción de humedales en pequeñas (20-2000 ha) cuencas agrícolas de la cuenca del Ebro (NE España). El modelo utiliza bases cartográficas existentes mejoradas (suelo y geomorfología), transformaciones geográficas simples (pendiente y distancia a cauces permanentes más cercanos) y otra cartografía creada (usos del suelo). Son necesarias escalas detalladas (~1:5000 y <30 m de lado de píxel) para trabajar con cuencas pequeñas. Un profundo conocimiento de la zona de estudio es necesario para reducir la subjetividad asociada a la decisión de experto. Los casos de estudio probaron que el 38% del área de las cuencas era adecuado par la restauración de humedales, del que un 15% era muy adecuado. En las cuencas grandes (>150 ha) la superficie adecuada

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* Moreno, D., Mander, Ü., Comín, F.A. and Pedrocchi, C. *Ecological Modelling*.Submitted.

ocupada la mitad que en las cuencas medias (60-150 ha) y pequeñas (<60 ha). La mayor parte de la superficie adecuada estaba ubicada en las partes bajas de las cuencas examinadas en el estudio. Hay una cantidad suficiente de zonas muy adecuadas para cumplir los requerimientos de construcción de humedales de flujo superficial para mejorar la calidad del agua. El modelo es una herramienta simple y útil para la planificación ambiental en áreas degradadas por la agricultura intensiva.

Introduction

Wetlands (re)construction is a common activity with many different purposes, such as wildlife preservation, water quality improvement, flood regulation or soil amelioration (Mitsch and Gosselink, 2000b). Many problems have emerged since there is little knowledge on the evolution of created or restored surface flow wetlands (SFW) after their construction. An understanding of natural processes taking place in future wetland locations, especially in those where new wetlands will be constructed, is essential to ensure the success of mitigation activities (Zedler, 2000; Eades et al., 2005). A large number of environmental parameters have been included in suitability and prioritization analysis. Among these, it highlights the use of the land-cover layer since the earlier studies on site selection for wetland restoration (Harris and Olson, 1997; Russell et al., 1997), and it is present in several studies from the last decade (Richardson and Gatti, 1999; Palmeri and Trepel, 2002; Van Lonkhuyzen et al., 2004; Newbold, 2005; White and Fennessy, 2005; Lesta et al., 2007). In these studies, other commonly used layers are soil, slope, elevation, historical wetlands and/or hydrology.

GIS-based suitability analysis resulted in a useful tool for landscape and environmental planning at regional (Baban and Wan-Yusof, 2003) and catchment scales (Wang et al., 2004; Saroinsong et al., 2007). More specifically, suitability and prioritization studies for wetland restoration and planning have had two main perspectives: wildlife preservation (Harris and Olson, 1997; Van Lonkhuyzen et al., 2004; McCauley and Jenkins, 2005) and water quality improvement. In the area of the former perspective, studies have focused on the retention of nutrients from agricultural non-point pollution at catchment scale (Trepel and Palmeri, 2002; Newbold, 2005; Lesta et al., 2007) and on the retention of sediments from upstream agricultural areas (Richardson and Gatti, 1999). Most of these studies have been performed in large agricultural catchments (>5000 ha), and therefore little is known about the application of the suitability models on small catchments (20-2000 ha), which is a handle size for implementing direct measures of wetland creation or restoration. It is also important considering a multipurpose perspective on wetland restoration planning (Knight, 1992a; Comín et al., 2001), especially in areas degraded by intensive agricultural use, where a few natural ecosystems remain as scattered and small patches in a very homogeneous landscape (Moreno et al., 2007).

The aim of this study is to demonstrate a method of landscape analysis that can be used to estimate the catchment suitability for the construction of SFWs and to present the results of this analysis in the agricultural catchments of the Ebro basin (Spain). The suitability assessment of wetland construction and potential patterns of wetlands will be performed in these catchments.

Study area

Monegros is a 270,000 ha inland semi-arid Mediterranean region (average annual temperature 14.5ºC, average annual precipitation 400 mm) with high interannual variability (Comín and Williams, 1993; Moreno et al., 2007) located in the centre of the Ebro River basin in NE Spain (Fig. 1). Soils are mainly dominated by a Tertiary structure composed of mudstones with different salinization levels (conductivity ranges between 1-10 $mS \cdot cm^{-1}$). If no quaternary layers were deposed upon this structure, they were classified as Orthic Solonchanks or Solonetzs. If no soil development was present and bared rock with tiny depositions was present, they were Calcic Lithosols. After a weak development of the soil and a small contribution of colluvial material, they were classified as Calcic Xerosols. When depositions of colluvial materials appeared widely, they were classified as Calcic Cambisols or Calcaric Regosols, and if these depositions are alluvial, they are called Calcaric Fluvisols. The landscape is severely altered and is composed of small plateaus with valleys in between, some gentle hills and low-altitude mountain chains (~800 m.) Most of this land was transformed into irrigated agricultural fields from the 1950s to the 1990s. Maize, alfalfa and cereal are the most common crops. Soil salinization and the abandonment of agriculture are now widespread in many parts of this region. As a consequence, the original semi-steppe vegetation, dominated by rosemary (*Rosmarinus officinalis*), thyme (*Thymus* sp.) and halophytes (*Salicornia* sp.), as well as perennial grasses (*Lygeum spartium*, *Brachypodium retusum*), was transformed into a landscape dominated by halophytic species (*Tamarix africana*, *Atriplex halimus*, *Suaeda vera*) in abandoned dry elevated zones and helophytes (*Phragmites australis*, *Typha latifolia*, *Scirpus holoschoenus*, *S. maritimus*, *Carex divisa*) in the abandoned wetter and lower zones of the valleys. In most of these zones, wetlands with permanent or intermittent surface water are formed and colonized by *Phragmites australis*, which is spreading and increasingly dominating the plant community because of its tolerance to changing water levels and salinity (Lissner et al., 1999a).

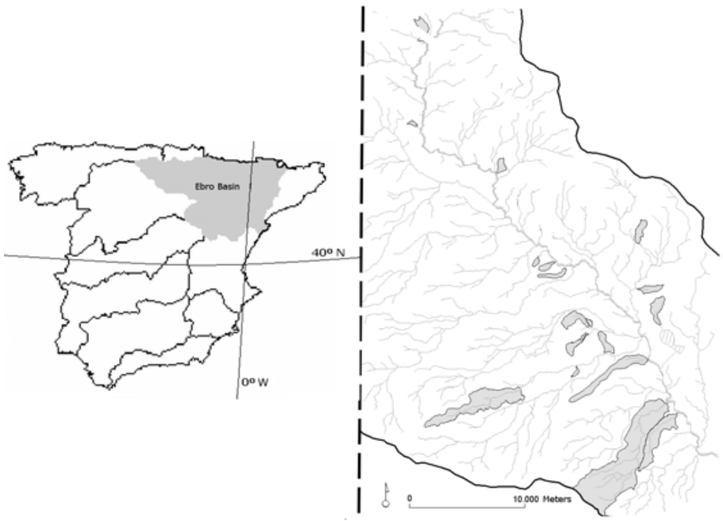

Figure 1. Location map of the study area with the main basins of the Spanish Iberian Peninsula (left). Selected subcatchments are shaded in grey. Subcatchments are included in the Flumen catchment and its drainage network (right).

Eighteen catchments within an 80,000 ha zone of the Monegros region were selected for this study (Fig. 1). All catchments with wetlands larger than one hectare (n=8) were included (maximum size was ~2,500 ha). Other catchments (n=10) with smaller wetlands (0.1–1.0 ha) were included if they were topographically similar to the catchments with larger wetlands, similar catchment area, and were accessible. All catchments were dominated by arable land, (77.36±13.61% of the catchment area), especially irrigated farmland (~73%). Rain-fed farmlands were only important (>25% of the catchment area) in two catchments (Moncalver and Paúles). Few patches of natural vegetation (17.01±10.44% of the catchment area) remained, and these included ~3% of spontaneous wetlands. The wetlands located in the catchments received agricultural wastewater coming from irrigation surpluses through drainage channels or via groundwater and flowing into the fluvial system. The hydrological system was entirely dependent on the system for the irrigation of fields in the surrounding area. Wetland vegetation grew spontaneously after transformation into irrigated farmlands, except for one wetland (Sangarrén), which is mostly fed by groundwater. Wetland location varied from hillsides with wastewater outcrops to flat valley floors.

Approaches and methods

Expert decision in model approach

A GIS-based suitability analysis was performed to identify potential locations for SFW creation. The model is designed to help (landscape) planners to create new wetlands in a simple, economic, realistic and ecological way. In devising this model, four categories were considered for decision-making: economy, hydrology, geology and ecology, and five data layers were synthesized: slope, land-use, geomorphology, soil and distance from permanent streams (Table 1).

Table 1. Used and initial cartographic data layers.

Theme	Category	Initial data layers	Initial resolution	Final resolution
Slope	Economy, hydrology	Digital elevation model	20 m pixel-side	3 m pixel-side (interpolated)
Land use	Ecology, economy	Aerial photographs	0.5 m pixel-side	1:5,000
Geomorphology	Geology, hydrology	Regional geomorphological map	1:50,000	1:5,000
Soil	Geology	Regional soils map	1:50,000	1:5,000
Distance to streams	Economy, hydrology	100 m multiple buffer from permanent streams observed in the field	—	1:5,000

Slope is a decisive factor in SFW construction. The main design considerations and construction budgets rely on it. The lower the slope, the cheaper the construction project, as a consequence of the smaller amount of earthwork. Extensive earthwork also involves higher energy expenses and, consequently, higher CO_2 emissions. Irrigated agricultural farmlands need flat areas when they are watered by flooding and gentle slopes (<10%) when sprinkling is used. There is a greater need for wetlands in areas with more farmland, which is usually found on flatter terrain.

The type of land-use determines the suitability of an area for the creation of wetlands. The most suitable areas would be those affected by irrigation, where the permanency of watering ensures water for wetlands. Areas with low human activity (abandoned, fallow or rain-fed farmlands) are also interesting for wetland construction, because it is expected that there will be a longer permanence of initial landscapes. Areas highly affected by human activities (channels, farms or urban areas) might be unsuitable as mitigation sites. These areas are usually affected by frequent impacts of human activities, and consequently the permanence of initial land-uses is less likely.

Geomorphological processes may point out areas where SFW creation is very unwise. The permeability of glacis and colluvial or mixed (alluvial/colluvial) deposits makes it impossible to build a wetland on them without impervious artificial layers (e.g. polyethylene sheet). This makes mitigation projects much more expensive. Also, the permanence of water on the soil surface is lower in such permeable geological formations. High terraces and monoclinal relieves are usually associated with ancient erosion processes that have moved permanent streams away from them. Low terraces and alluvial deposits near rivers are suitable for the placement for wetlands because of the limited construction work required and the proximity of permanent streams.

Soil properties such as texture, depth, composition or development affect future potential mitigation sites because they facilitate or hinder the establishment of natural vegetation and water storage. Work with relatively undeveloped soils such as Lithosols or Xerosols is often mechanically problematic due to their thin layer of real soil. Regosols are weakly developed soils and, in our study area, have accumulations of gravels, which make them very unsuitable for water storage. Cambisols are more developed and have a cambic horizon, which makes them permeable and unsuitable for water accumulation. Solonetzs and Solonchaks are dominated by salinization and sodification processes respectively. Our study area is mostly composed of different proportions of clay and silt. The mixture of salts, clays and silt produces an effect of loss of structure (very unsuitable for agricultural use) and a high capacity for water storage. Finally, Fluvisols are interesting soils for wetlands because of their proximity to streams and the consequently high water table.

Water availability is a decisive factor in selecting SFW location. The distance from permanent streams has a strong influence on the total cost of the mitigation project. Long distances imply high energy requirements and financial expenditures. 150-meter rings were considered to be the spatial units that would noticeably increase energy requirements and financial expenditures. Suitable areas were found within the first ~500 meters from a permanent stream. We considered unsuitable for wetland mitigation all catchment area located above the highest point where water ran permanently. In this area, only water pumping could keep water on the wetlands, and this energy requirement and financial expenditure was considered to render wetland creation unfeasible. Moreover, pumping involves high maintenance costs.

Table 2. Suitability classes of the land-use, geomorphological and soil types of the study area.

Land-use	Geomorphology	Soil	Suitability
	Monoclinal relieves Glacis	Xerosol calcic Regosol-Cambisol calcaric	-3
Channel Farm Urban	Alluvial-colluvial deposits High terraces	Cambisol-Litosol calcic	-2
Arboreal vegetation			-1
Erosion desert Herbaceous/scrub vegetation	Erosion surfaces Low intense geomorphological process	Cambisol calcic	0
Fallow land		Solonchak orthic	+1
Abandoned land Rain-fed land		Solonetz orthic	+2
Irrigated land Wetland	Alluvial deposits and low terraces	Fluvisol calcaric	+3

Cartographic analysis

Five themes were synthesized to build the suitability map (Table 1). The slope was calculated using a 20 m pixel-size digital elevation model (DEM) and interpolated to 3 m pixel-size. We calculated slope as the maximum rate of change in height between each cell and its neighbours (ArcMap 8.3, ESRI Inc). Land-use was estimated using aerial photographs with a resolution of 0.5 m taken in 2003. Eleven land use types (irrigated farmland, rain-fed farmland, fallow land, abandoned land, stocks farms, arboreal vegetation, dry shrubland-grasslands, erosion deserts, wetlands, irrigation channels and urban areas) were digitalized in all catchments using ArcGIS 8.3 (ESRI, Inc.). The land cover types in the photographs were checked *in situ* by direct observation, and their areas were estimated. Both geomorphological themes and soil themes were synthesized from existing regional maps created by the Aragon Government at 1:50,000 scale. Using these maps and aerial photographs, the scale was improved to a scale of 1:5,000. Soil and geomorphological unit types were also checked *in situ* by direct observation, and their areas were estimated. The distance from permanent streams was calculated from digitalized streams and the buffer function (150 m distance rings) of ArcMap 8.3 (ESRI Inc.). Every stream was recognized in the field by direct observation and pointed out on an aerial photograph at 0.5 m pixel-size. Streams were considered to be permanent when water was observed to be running on at least three of the four visits carried out annually during the years 2004 to 2006. The contour line passing through the highest point of the permanent stream was considered to be the maximum height for wetlands construction suitability in the catchments. Vector-based data layers were then converted to raster-based layers using the Spatial Analyst function of the ArcMap 8.3 program (ESRI Inc.) with a 3 m pixel-size resolution.

Table 3. Distribution of suitability for wetland creation in the catchments studied.

Catchment	Area (ha)	Suitable (%)	Very suitable (%)	Suitable + very suitable (%)
Torres	21.89	13.28	8.72	22.00
Af. Filadas	22.51	14.53	17.56	32.09
Poleñino	28.46	6.57	8.41	14.97
Capdesaso	54.73	17.32	12.22	29.54
Chamarcal	58.06	7.46	23.64	31.10
Grañén	59.07	6.66	91.98	98.64
Barranquetes	80.61	10.58	12.08	22.66
Balsas	89.94	36.91	17.85	54.76
Río Ancho	102.20	26.84	40.02	66.85
Planos	108.36	32.63	21.63	54.26
San Lorenzo	111.94	62.90	19.81	82.71
Af. Val de Zgz	112.82	16.22	11.25	27.47
Sangarrén	147.34	45.93	7.17	53.11
Sariñena	433.83	16.00	8.37	24.37
Matical	436.73	16.85	14.62	31.47
Albalatillo	521.66	25.04	11.55	36.60
Paúles	965.26	7.78	3.70	11.48
Moncalver	2535.46	26.10	4.95	31.05

The types of each data layer were divided into classes, and one value was assigned to each class. The values varied from -3 to +3, which represented the lowest and highest suitability respectively (Table 2). A negative value involved unsuitability and a positive one suitability. A raster map was then synthesized by reclassifying with new values. Weights were associated to each data layer in regard to their importance in the selection of optimal wetland location. Water availability was considered to be the most significant condition for the rejection of a location for the placement of created wetlands, and the greatest weight (1.5) was assigned to the distance from permanent streams layer. The presence of large areas covered by glacis or other colluvial or mixed deposits was also a determinant in rejection for wetland creation, and an additional weight (1.25) was assigned to the geomorphological layer. The rest of the data layers continued to have a weight of one. To make suitability values easier to interpret, they were centred between -1 and +1 by dividing them by the maximum value of the resulting sum of the values of all data layers. The resulting data were discretized in intervals of 0.2. The final model was thus:

$$Suitability = \frac{1.5 * Distance + 1.25 * Geomorphology + Slope + Landuse + Soil}{Max\left(1.5 * Distance + 1.25 * Geomorphology + Slope + Landuse + Soil\right)},$$

where distance, geomorphology, slope, land-use and soil were the values of the date layers described in table 1 and assigned according to Table 2.

Results: Case studies

Suitability maps obtained using the model with the 18 selected catchments make up 38.44±18.56% (mean±SD) of the areas suitable for SFW creation (Fig. 1 and 2; Table 3). 14.99±8.62% were considered to be very suitable places (>0.4), and 23.45±15.11% were found to be suitable places (0-0.4) based on the model. Suitable area ranged from ~7% of Poleñino catchment to ~98% of San Lorenzo catchment. This range was from ~3% to ~91% for very suitable areas (Table 3). This very high range of variability between suitable areas was typical of medium-size catchments (60-150 ha; Table 4). In small (<60 ha; n=6) and large catchments (>150 ha; n=5), the variability between suitable areas was lower (CV<40%) than in medium catchments (CV>60%; n=7). Figure 2 illustrates how widely the dark area (suitable area) varies from one catchment to other in medium and small sized catchments, and figure 3 shows how the dark area is mainly located in the lower third of large catchments.

Table 4. Proportion (mean±sd) of catchments suitable to host created wetlands regarding to the main size classes. Proportions are referred to the total catchment.

Area (ha)	n	Suitable (%)	Very suitable (%)	Suitable + very suitable (%)
<60	5	28.79±9.92	16.09±13.04	44.89±15.50
60-150	7	22.43±20.49	15.61±6.40	38.04±23.61
>150	5	13.29±4.79	9.43±1.87	22.72±6.05

Model adjustment and discussion

The GIS modelling approach proved useful and simple for the location of SFW creation sites at catchment scale, and therefore was an interesting tool for agricultural landscape planning. The construction of SFWs for the improvement of water quality and the strengthening of biodiversity in locations selected by the suitability model ensure low cost and energy expenditure. The use of easily accessible data layers and the simple transformations they require (slope and distance from a permanent stream) make the model widely useful. The importance of using easily accessible information by administrative officials with standard knowledge of GIS-software in landscape planning was also considered in previous models as essential to extend its use (Palmeri and Trepel, 2002; Lesta et al., 2007). It is expected that the accessibility of improved and more complete geographical information will increase in the future (Van Lonkhuyzen et al., 2004).

More precise layers (~1:5000 or 3-5 cell size) will be needed for small catchments (<2000 ha) if high accuracy is desired. Original information usually has rougher scales (<1:50,000 or 30x30 m cell size) (e.g. Russell et al., 1997; Richardson and Gatti, 1999; Palmeri and Trepel, 2002; Lesta et al., 2007) and produces useless maps in small catchments. Greater accuracy than that used in this study (cell size=20 m) was also needed in DEM when working with small catchments.

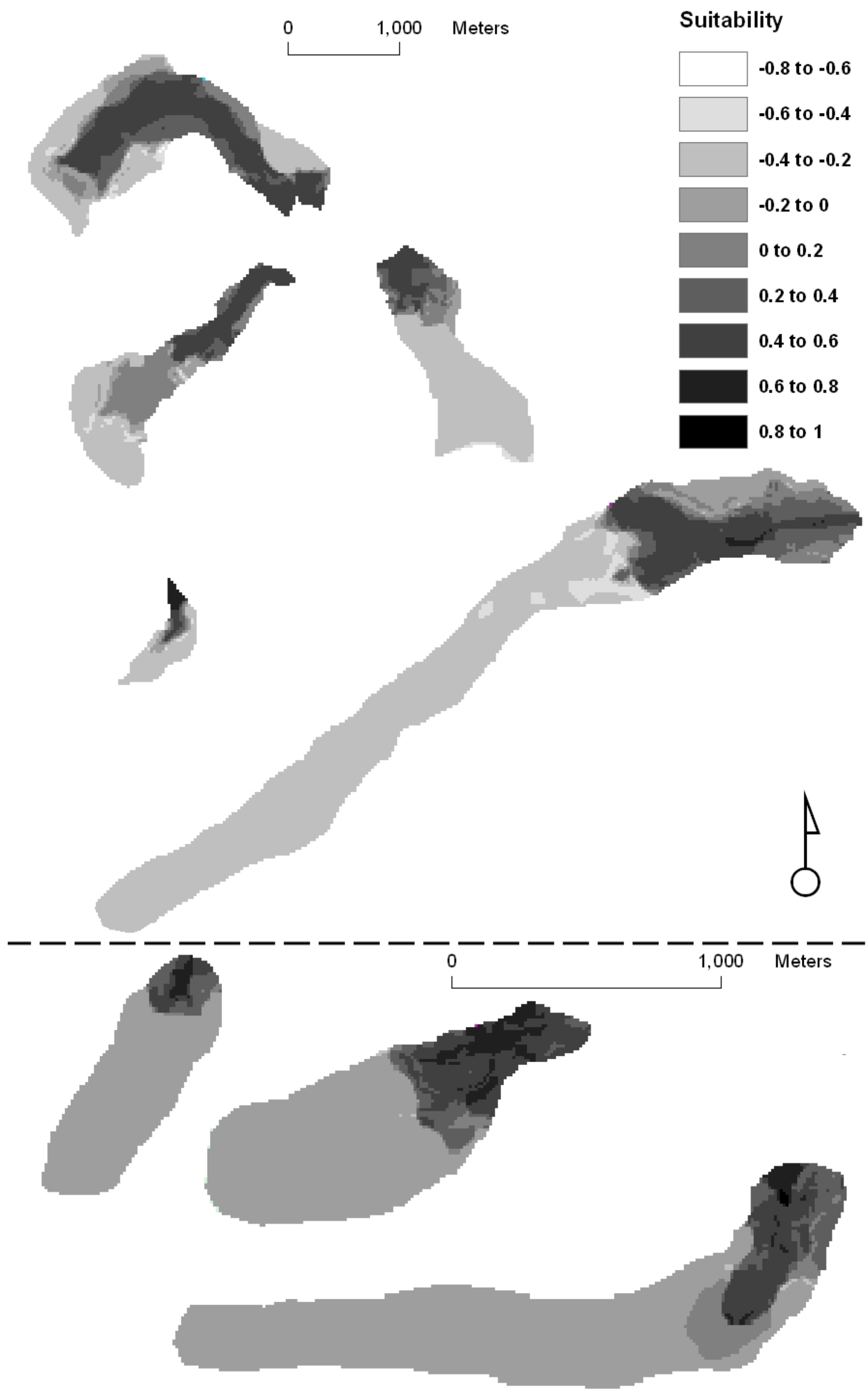

Figure 2. Results of the suitability analysis of some of the small (<60 ha) and medium (60-150 ha) sized studied catchments.

The assignment of suitability values and weights needs support from experts' decisions. The inherent subjectivity associated with this phase of the analysis was noted by other researchers. Baban et al. (2003) reported that GIS methodology makes decision-making processes more objective. We propose that a deep knowledge of the region under analysis is essential to reduce associated subjectivity and, subsequently, to improve the usefulness of the model in site selection (Van Lonkhuyzen et al., 2004). The model must be seen as a flexible tool that is adaptable to different conditions of study areas that are thoroughly known (Zedler, 2000; Van Lonkhuyzen et al., 2004). We consider the model to be a complementary tool that must be integrated into decision-support models used in environmental and agricultural planning (Wang et al., 2004).

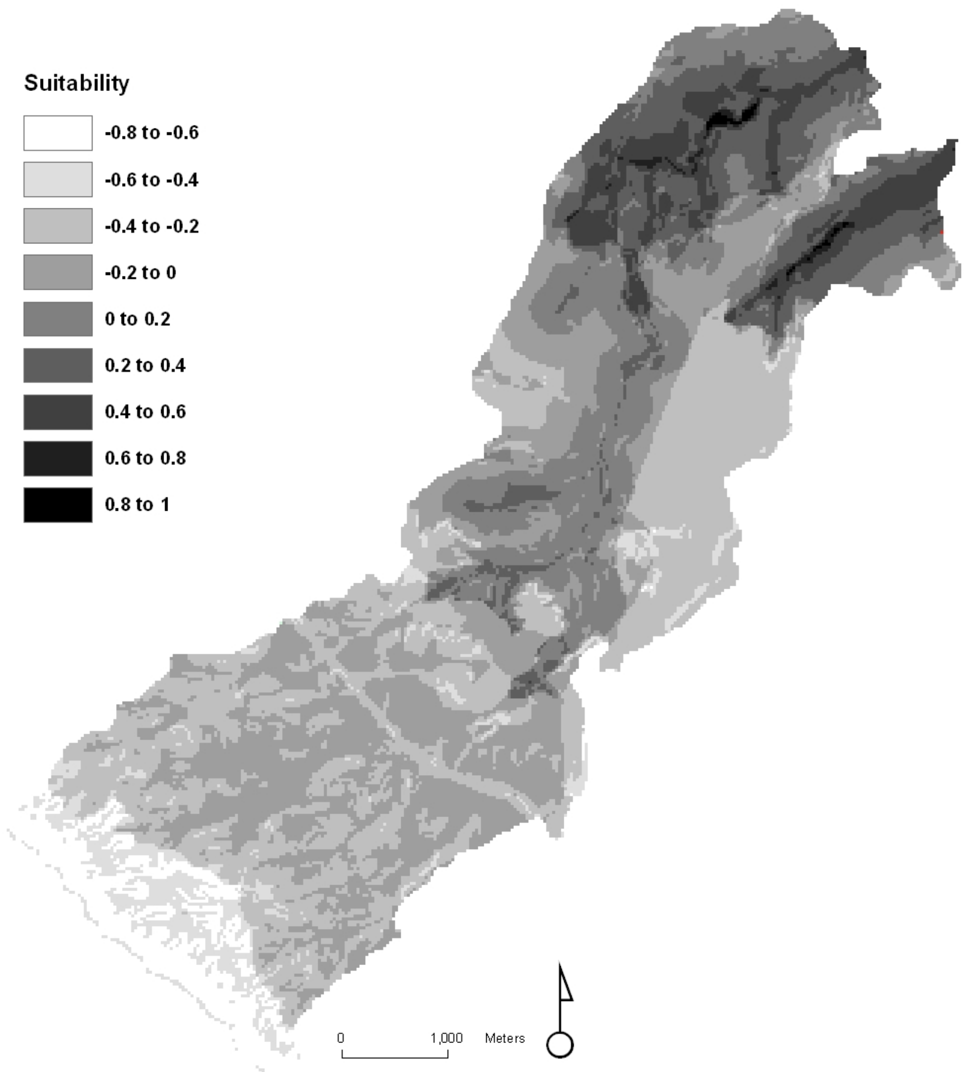

Figure 3. Results of the suitability analysis of some of the large studied catchments (500 and 2500 ha).

The application of the model to the studied catchments provided high rates of areas suitable for SFW construction (38%), including 15% of very suitable areas. These results are very similar to those reported by Lesta et al. (2007) for several Estonian counties, where 42% of their area was suitable for wetland construction, including 16% of very suitable areas.

More restrictive models recommended between 8.0 and 14.6% of a large (2197 km^2) agricultural catchment in Ohio (White and Fennessy, 2005) or 11% of a 40 km^2 catchment in Italy (Palmeri and Trepel, 2002) as the most suitable areas for wetland restoration. These restrictive percentages are not far from those proposed by Lesta et al. (2007) and this study as very suitable areas, and consequently were the first to be selected for wetland construction. The restoration model for riparian wetlands proposed 2.29% as a high priority for wetland restoration (Russell et al., 1997), which was partly due to the fact that only the floodplain area was considered in the model. The amount of very suitable area for wetland construction found in this study closely matches the area of wetlands needed at catchment scale to remove most of the nitrate from agricultural runoff (3.25-5.60%) proposed in the same study area in previous studies (Moreno et al., 2007).

The model showed less variability in the amount of suitable area in large (>150 ha) and small (<60 ha) catchments. The suitable area in large catchments was almost half that in the small and medium catchments. This is because most of the permanent streams fed by irrigated fields were concentrated in the lowest parts of the valleys where the terrain was also flatter. This reason, together with the fact that in some medium and small catchments wastewater outcropped from permeable stratus in an intermediate point between the head and the mouth, meant that most of the suitable area for wetland construction in these catchments was also concentrated in the lower parts.

Using this model in decision-making processes in landscape and agricultural planning will make it possible to easily determine the most suitable areas for SFW construction in new agricultural developments and in the modernization of old ones. The model shows that there is enough "very suitable" area to construct all needed wetlands for water quality improvement, and they must be concentrated in the lower parts of the small catchments (20-2000 ha). This will result in a mosaic of constructed wetlands at regional or large watershed scale and an integrated landscape in areas that have been degraded by intensive agricultural use. This may satisfy the combined objectives of improving water quality and strengthening biodiversity.

Implications for practice

- Basis cartography (soils, geomorphology, DEM, land-uses and permanent streams layers) is needed to identify suitable location for wetland construction.
- Restoration of small catchments (<2000 ha) requires high resolution cartography (1:5000 or 3-5 m cell-size) to ensure a detailed approach to real conditions.
- Expert decisions must be supported by deep knowledge of the study area
- Low-cost actions must be implemented in lower parts of small catchments (<2000 ha) where wastewater flow is concentrated
- As an indicator, 15% of the area of small catchments is very suitable for wetland construction activities in semi-arid conditions.

Acknowledgements

Special thanks are given to Merje Lesta from the University of Tartu for her useful comments. This work was supported by CICYT (REN2003-03040), Aragon Government –D.G. Investigación, Innovación y Tecnología– Research Group on Ecological Restoration (E 61), and DGA (CONAID)–CAI (CA1/07), Estonian Science Foundation grant No. 6083, and Target Funding Project No. 0182534s03 of the Ministry of Education and Science of Estonia.

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Chapter 8

Integrating multiple objectives for wetland restoration and construction planning*

Abstract

Interest has recently been increasing in new objectives of restoring and constructing wetlands (landscape integration, soil improvement or carbon sequestration) that are different from classical objectives, such as improving water quality, biodiversity, and flooding control. A significant base of knowledge already exists on different objectives of wetlands restoration. Recent interest has also developed in restoring wetlands with multiple objectives. To attain this goal, it has been necessary to understand the conflicts and compatibilities of these restoration objectives. This level of understanding will guide us in determining whether we wish to create one wetland with multiple objectives or several wetlands with different objectives. Afterwards, a restoration strategy can be defined by establishing a hierarchy of restoration objectives and determining the scale of restoration (site, catchment, basin or territory). In general, the larger the scale, the more suitable the implementation of different restoration objectives, with the catchment and especially basin scales are considered optimal for multiple objectives. A deep knowledge of local limitations will significantly increase the likelihood of success. Following these guidelines, restored and constructed wetlands would be more similar to natural wetlands with more ecological functions, especially if spatial distribution is taken into account during restoration planning. Restored wetlands could then provide an array of integrated services adapted to local ecological and social needs.

Resumen

El interés en restaurar y construir humedales con nuevos objetivos (integración paisajística, mejora del suelo o secuestro de carbono) ha crecido recientemente en relación con los objetivos clásicos, como eran mejora de la calidad del agua, biodiversidad y control de avenidas. Actualmente, existe una significativa base de conocimiento sobre diferentes objetivos de restauración de humedales. También existe un reciente interés en restaurar humedales con múltiples objetivos. Para lograr esta meta, es necesario conocer los conflictos y compatibilidades de esos objetivos de restauración. El nivel de conocimiento guiará en la necesidad de crear un humedal con múltiples objetivos o varios humedales con diferentes objetivos. Previamente, se puede definir una estrategia de restauración

* Moreno, D., Comín, F.A. and Pedrocchi, C. Under internal review.

estableciendo una jerarquía entre los objetivos de restauración y determinando la escala de la restauración (puntual, pequeñas cuencas, grandes cuencas o territorio). En general, cuanto mayor es la escala más adecuada es la consecución de diferentes objetivos de restauración. La escala de cuenca y, especialmente, la de gran cuenca se consideran óptimas para la restauración con objetivos múltiples. Un profundo conocimiento de las limitaciones locales incrementará significativamente las posibilidades de éxito. Siguiendo estas directrices, los humedales restaurados y construidos serán más parecidos a los naturales, con más funciones ecológicas, especialmente, si se tiene en cuenta la distribución espacial durante el proceso de planificación. Así, los humedales restaurados pueden proveer una serie de servicios integrados adaptados a las necesidades ecológicas y sociales locales.

Introduction

Loss or destruction of wetlands has entailed the loss of valuable services (water supply, food or raw materials) all around the world (Mitsch and Gosselink, 2000; Assessment, 2005). These losses have reached up to 50% of wetland area in Europe and North America, especially in the last century, basically as consequence of agricultural transformations, infrastructure development, water withdrawal, overexploitation and the introduction of alien species (Assessment, 2005). Recently, wetland restoration or construction has received increased attention because of the multiple services performed all around the world (Zedler and Kercher, 2005). As wetland services become more highly valued (Costanza et al., 1997), the worldwide number of restoration and construction projects increases each year. Classical perspectives of wetland restoration have included water quality, biodiversity, flood control or recreation (Knight, 1992; Zedler, 2000). Although these still remain the most relevant objectives, new objectives are emerging such as maintaining landscape heterogeneity, fisheries preservation, and carbon sequestration (Comín et al., 2001; Table 1).

Interactions between restoration objectives have barely been studied. Several relevant studies have demonstrated that what is good for one objective, like N removal or biodiversity strengthening, may be bad for another, like P removal (Hansson et al., 2005); or that which is good for one animal group (birds) may not be good for another (fish), indicating that one restored site cannot maximally serve all potential functions (Findlay et al., 2002). To implement restoration objectives, several arrays of guidelines have been provided, mainly based on ecological knowledge (Zedler, 2000; Zedler, 2005), hydrological science (Acreman et al., 2007) or landscape limitations (Arheimer et al., 2004; Newbold, 2005; Lesta et al., 2007). Some indications have also been documented about the optimal scale of restoration (Kershner, 1997; Zedler, 2003) and about restoration strategies (Thom et al., 2005). However, the field lacks an integrative approach for restoring or constructing wetlands under local limitations at a specific scale.

To observe the evolution of the different perspectives of wetland restoration or creation projects, the number of studies over the last 30 years found in journals included in the Science Citation Index Expanded (ISI Web of Science, Thompson Corp. 2008) has been counted for four relevant purposes. A brief bibliographic review will summarize which wetland objectives have been historically, and are currently, the most valued objectives of wetland restoration, along with their current state of knowledge. The aim of this study is to provide feasible guidance on how to restore or create wetlands with multiple objectives (water quality, biodiversity, landscape, etc.) in catchments degraded by intensive agricultural uses under Mediterranean semiarid conditions. Some specific advice is given on how to restore or construct wetlands under semi-arid Mediterranean conditions to provide some valuable services according to local need.

Perspectives on wetland restoration

Four purposes for wetlands restoration were selected due to their importance in the restoration of ecosystems degraded by intensive agricultural use in a semi-arid Mediterranean area. A detailed description of the area is reported in Moreno et al. (2007). These ecosystems were characterized by low water quality in streams, high landscape homogeneity, low biodiversity, and degraded soils. These four factors were previously studied in the semi-arid Mediterranean area, and they were considered as priorities for restoration in this study (Table 1). The first perspective was water quality improvement (including nutrients, solids, salts, heavy metals and organic pollutants). In order to select only wetlands where conditions were close to natural conditions, this literature review was restricted to articles related to restored or created wetlands classified as "surface flow wetlands" (SFW; Kadlec and Knight, 1996) on natural soils. The second perspective was the strengthening of bird populations in restored or created wetlands. Articles that studied any aspect of the bird populations established on restored or created wetlands were included. The third perspective was the study of wetlands at landscape scale. All articles where the role of wetland creation or restoration at landscape scale was studied were reviewed. The fourth perspective was the improvement of the soil. Articles that studied processes that lead to an improvement of the main features of natural soils (organic matter, bulk density, salinity, inorganic nutrients, heavy metals and organic pollutants) were included.

Improving water quality

Table 3 shows that 51% of the articles were related to water quality improvement. This is one of the classical objectives for wetland creation or restoration. Although there were old and complete studies not selected by the search criteria (e.g. Jones and Lee, 1980; Hammer, 1989), the first study found in SCI journals was published in 1989. Since then, the volume of studies has increased dramatically each year through the present (Table 2). This is mainly due to the strong interest shown in nutrients retention (N and P), which is especially interesting in agricultural catchments to treat agricultural runoff or sewage from rural areas (Kuusemets and Mander, 1999; Mander et al., 2000; Zedler, 2003; Novotny, 2005). These methods have been proven useful in most conditions, such as arid, boreal, temperate or tropical, retaining widely variable amounts of N (30-99%) and P (0-99%) (e.g. Romero et al., 1999; Woltemade, 2000; Comín et al., 2001; Koskiaho et al., 2003; Kantawanichkul and Somprasert, 2005; Moreno et al., 2007). Surface flow wetlands on natural soils are indeed interesting because of their low cost of construction and maintenance, which has favoured the quick increase in their number worldwide (Hammer, 1992; Mitsch and Gosselink, 2000; Kovacic et al., 2006). Recently, it has been demonstrated that the role of wetlands in the emission of greenhouse gases (Mitsch and Gosselink, 2000; Verhoeven et al., 2006) must be assessed before starting any new project of wetland restoration or construction.

Table 1. Observed results and design and management implications of some studies with different perspectives carried with constructed and existing wetlands in areas degraded by intensive irrigated agricultural use under Mediterranean conditions.

Target	Observed benefits	Observed fails	Design and management implications	Publications
Water quality improvement	99% N retention Reduction of salts release (down to 9% in four years)	P release (0.43-2.99 $gP \cdot m^{-2}$) Salts release (67-244 $gNa \cdot m^{-2}$) C release (~25 $mgCaCO_3 \cdot l^{-1}$)	~5,000 m^2 area Four days of WRT Permanent flooding 3.5-6.5% of catchment to wetlands	Romero et al. 1999 Comín et al. 2001 Moreno et al. 2007 Moreno et al. 2008a
Soil properties improvement	~70% NO_3-N reduction ~35% Av. P reduction ~70% Salts reduction ~70% SOM increase ~100% TOC increase	Slow accumulation of SOM and TOC (0.03 to 0.5% per year) Salts release	Permanent flooding Long-term perspective Mosaic in the catchment	Moreno et al. 2008b
Biodiversity strengthening	Bird diversity increase Bird richness increase Bird abundance increase	Uncontrolled burning reduce bird population diversity and abundance	>0.6 ha Irregular shapes Promote different plant communities (reedbeds and bushes) Avoid uncontrolled burnings Scattered in the landscape	Comín et al. 2001 Moreno et al. 2008c
Landscape improvement (landscape heterogeneity)	Landscape heterogeneity increase	Low overall landscape diversity (0.72)	Current diversity appear in lower parts of catchments	Comín et al. 2001 Moreno et al. 2007
Landscape improvement (water quality improvement)	N and P reduction Salts reduction	Small amount of existing wetlands Low functionality of existing wetlands	Increase the current area of the catchment covered by wetlands (~3%) Work in small catchments Compact and flat bottom Scattered and numerous Improve existing seeping areas	Moreno et al. 2008d
Landscape integration (optimal location)	Low-cost construction Landscape integrated Optimize functionality Simplicity of the model	Small catchments (<2000 ha) always requires detailed field observation	Work in lower parts of catchments Work in small catchments (<2000 ha) 15% of the catchment is very suitable	Moreno et al. 2008e

Table 2. Number of SCI articles dealing with the study perspectives and published in selected journals* during the last 30 years.

Year	n	Water quality	Bird populations	Landscape scale	Soil features
Before 1990	12	1	0	0	0
1990-1994	5	3	2	1	1
1995-1999	5	16	9	6	2
2000-2004	5	32	10	10	10
2005-2007	3	32	11	11	9
Total	30	84	31	28	22

* Reviewed journals: Restoration Ecology, Ecological Engineering, Ecological Applications, Ecological Modelling, Journal of Applied Ecology, Trends in Ecology and Evolution, Frontiers in Ecology and the Environment, Environmental Management, Journal of Environmental Management, Soil Science Society of America Journal, Journal of Environmental Quality, Water Resources Management, Water Science and Technology, Hydrology and Earth Systems Sciences, Landscape and Urban Planning, Biological Conservation, Chemosphere, Science of the Total Environment, Wetlands, Journal of Soil and Water Conservation and Journal of the American Water Resources Association.

The landscape scale

The jump to a landscape scale in wetland restoration or construction mainly took place in the late 90s, and today, the landscape perspective only represents 17% of all selected papers (Table 2). However, this perspective is growing quickly because of the interest in integrating wetland restoration and construction in catchment or regional planning (e.g. Kuusemets and Mander, 1999; Mitsch and Gosselink, 2000; Zedler, 2003; Arheimer et al., 2004; White and Fennessy, 2005; Montreuil and Merot, 2006). One of the best-studied topics dealt with by this recently developed perspective is the amount of restored or constructed wetlands necessary to ensure water quality improvement. At a large scale, Mitsch et al. (2000) proposed to leave 1% of the Mississippi basin watershed as wetlands in order to remove 40% of nitrate-N, and Larson et al. (2000) proposed 5% to remove 30-50% of nitrate-N in Illinois, a similar percentage to that recommended by Arheimer and Wittgren (1994) in Sweden. At a small scale, Hammer (1992) proposed the creation of wetlands in 2% of northern Maine to almost remove all N, and Moreno et al. (2007) advised the construction of wetlands in 3-5% of the agricultural catchments in the Ebro basin of Spain. Other topics studied included the use of wetlands to improve the heterogeneity in landscapes degraded by agricultural uses (Comín et al., 2001; Moreno et al., 2007). They demonstrated that the use of wetlands could reduce landscape homogeneity. Finally, a third topic studied was the optimal location of these restored or constructed wetlands in a region or catchment. Several models based on GIS cartography have been reported to find the best locations in agricultural catchments in order to improve water quality

(Palmeri and Trepel, 2002; White and Fennessy, 2005; Richardson and Hussain, 2006; Lesta et al., 2007) or in regions to strengthen wildlife habitats (Van Lonkhuyzen et al., 2004), giving different weights to GIS layers (land-uses, vegetation, hydrology, soils, slope, etc.) depending on the particularities of the study area.

Creating habitats for birds

The evolution of the study of bird communities in restored or constructed wetlands followed a similar trajectory to those of the landscape, and they have only recently been studied in depth. These represent 19% of all studies reviewed (Table 2). Some of these articles studied the influences of the restoration of wetlands on endangered or rare species of birds, such as European bitterns (*Botaurus stellaris*) (Gilbert et al., 2005) or Red-crowned cranes (*Grus japonicus*) (Prato et al., 1995). The colonization or use of restored wetlands by birds has been studied in tidal marshes for waders (Armitage et al., 2007) and in coastal lagoons for waterbirds (Comín et al., 2001). These studies reported a significant increase in the use of restored areas by birds (mainly for feeding) compared to the pre-restoration situation. In some cases, restoration of wetlands with their original features cannot mean an increase in their use by aquatic birds, and the possibility of leaving them in their current state must be considered. One of these cases has been illustrated by Takekawa et al. (2006). These authors observed that current abandoned evaporation salt ponds could be more intensively used by birds than the original tidal marshes existing after their transformation. Bird communities have also been used as indicators of success or failure in wetland reconstruction/restoration. Yang et al. (2006) considered a major success the permanent establishment of bird populations in wetlands (reedbeds) constructed to treat heavy metals from mining after 16 years of wetland functioning. Weller (1999) and Brawley (1998) compared bird use of a restored floodplain and a tidal marsh, respectively, with natural paired ecosystems to assess the success of the restoration projects. Sometimes, restoration projects do not follow the desired trajectories and fail. This happened in some restored salt marshes, where the endangered Clapper rail (*Rallus longirostris levipes*) did not find the necessary habitat to feed, and to consequently establish populations, which was one of the objectives of the restoration project (Zedler, 1993). Indeed, Zedler et al. (1999) stated that the desired trajectory of this restoration project for the Clapper rail will not be achieved in the short term.

Improvement of degraded soils

The real increase in the study of the soil in restored or constructed wetlands happened around five years later than it did for the rest of the approaches (Table 2). This represents the lowest percentage of the selected studies, at 13%. A lower number of studies on soil ecology regarding to other environs such as water, air or vegetation commonly happens because of the associated difficulties in soil sampling and analytical methods. The spatial variability of soil properties was one of the most widely studied factors on restored and constructed wetlands because of its decisive influence on the distribution, composition, abundance and size of all organisms present in the wetland (Gallardo, 2003). Restored or constructed wetlands take time to evolve towards the natural diversity of the soils of natural wetlands, but their variability has increased over the years (Bruland and Richardson, 2005; Mitsch et al., 2005). Another essential property of wetlands soils is their capacity to retain phosphorus from the water in their matrix by precipitation (Zurayk et al., 1997; Van den Berg and Loch, 2000) or immobilization by soil microorganisms (Silvan et al., 2003), but this property can vary and decrease with time, and must be periodically reevaluated (Hogan et al., 2004). The accumulation of organic matter is one of the most important indicators of soil quality, and its increasing decomposition rates in restored or constructed wetlands provide an indicator of their stage of evolution and functionality (Brooks et al., 2006; Bruland and Richardson, 2006).

New perspectives on wetland restoration

The improvement of the water quality in SFW on natural soils is not only the most studied topic, but also that which has experienced the most drastic increase in the number of articles published in SCI journals. It remains the subject of the greatest interest for the restoration or construction of wetlands. New perspectives, however, including those found in the literature, but not included in this study, have become more important, especially over the last 10 years (Tables 1 and 3). This indicates that the interest in restoring and constructing wetlands using more complex perspectives is increasing, whereby the functionality and integrity of these new wetlands will be closer to those natural, and the benefits obtained from their existence will be greater.

Table 3. Reference studies of some of relevant objectives in wetland restoration planning.

Restoration objective	Articles
Flood abatement	(Hey and Philippi, 1995; Zedler, 2003; Cox et al., 2006)
Carbon sequestration	(Whiting and Chanton, 2001; Mitra et al., 2005; Zedler and Kercher, 2005)
Fisheries preservation	(Koonce et al., 1996; Kaly and Jones, 1998; Richardson and Hussain, 2006)

Multipurpose restoration of wetlands

Only recently, wetlands have been restored or constructed with a multipurpose perspective, although ancillary benefits of wetlands constructed for wastewater treatment were previously discussed (Knight, 1992). In this early study, ecological functions (secondary production of fauna, habitat diversity and export to adjacent systems) and services to society (aesthetic, hunting or recreation) were discussed. More recently, some studies have dealt with wetlands restored or constructed with a multipurpose perspective. Comín et al. (2001) reported an improvement of the bird and landscape diversity in wetlands where N and P were also removed from agricultural runoff in a Mediterranean area. Hansson et al. (2005) studied in depth numerous conflicts between services produced by wetlands, focused on nutrient retention and biodiversity under boreal conditions. Among their findings, they highlighted that shallow and large wetlands with high shoreline complexities are likely to have high bird and macrophyte species richness and high nitrogen retention, whereas small, deep wetlands are likely to be more efficient in phosphorus retention, but less valuable if high biodiversity is desired. Findlay et al. (2002) stated that one site cannot maximize all potential functions, and demonstrated that functions for birds were negatively correlated with functions for fishes in the tidal freshwater wetlands of the north-eastern USA. Furthermore, prioritization will depend on the need of the habitat for every animal group in each region. Although more research is needed in wetland restoration with a multipurpose perspective, as indicated by Benyamine et al. (2004), an initial approach is carried out here by studying potential conflicts and compatibilities between restoration objectives, and by examining ways to integrate them or establish hierarchies on the basis of the existing information.

Compatibilities and conflicts between restoration objectives

Water quality improvement vs. biodiversity strengthening

Now, with the current knowledge of some objectives of restoration, we can ask whether it is better to restore with one or multiple objectives. We will see that the answer depends on the objectives we would like to attain and the local limitations of the area to be restored. The studies illustrated in table 1 under semi-arid Mediterranean conditions demonstrate how wetlands and landscape features may determine conflicts and compatibilities between high priority purposes of wetland restoration (Fig. 1). Relevant conflicts were found between water quality improvement and biodiversity strengthening. The area needed for water quality improvement (99% of NO_3-N reduction) is close to that needed for biodiversity strengthening, but this will suppose that all wetlands had almost the minimum size to strengthen biodiversity, thereby avoiding the development of more complex populations of birds linked to larger wetlands, where plant and ecosystem heterogeneity is higher (Craig and Beal, 1992; Weller, 1999; Moreno et al., 2008d). Landscape improvement for water quality improvement is favoured by wetlands dominated by homogenous and compact reedbeds (Moreno et al., 2008c), where communities of birds are generally poor. This supports the latter statement and reinforces the conflict between water quality improvement and biodiversity strengthening, because rich and diverse communities of birds need irregular patch shapes (Table 2). This situation was even more difficult in our study area if P retention was taken into account, because in their current state, the studied wetlands did not show a net retention of P. New designs could be used to overcome the conflict found by Hansson et al. (2005) between P retention and biodiversity support. On the other hand, the spatial distribution of restored wetlands in the landscape for water quality improvement and biodiversity strengthening must be scattered, which makes this approach compatible with catchment scale planning.

Soil improvement

For other combinations of purposes, although compatibilities could initially exist, occasional situations could happen which lead to a confrontation between them. In this sense, less restrictive conflicts could appear between water quality improvement and soil improvement (Fig. 1) when wetlands are restored or constructed upon soils affected by saline, sodium or calcareous conditions, as occurred in the area studied. Salts can be released to the water column, making its quality worse. This problem could be reinforced in situations where intermittent flooding occurs and salt release is favoured (Rodríguez-Ochoa et al., 1998). Also, $CaCO_3$ could be released from the soil to the water, becoming a source of C in the ecosystem (Moreno et al., 2008a). However, restoration with the purpose of water quality improvement, at wetland and landscape scales, at the proposed portions of the catchment (3-5%), in a scattered spatial distribution and in permanently flooded wetlands meet most of the recommendations for restoration with a soil improvement perspective (Table 2). Additionally, the scattered spatial distribution also meets the biodiversity strengthening perspective. Thus, this combination of purposes is recommended with some restrictions in saline soils.

	Landscape integration[a]	Landscape improvement[b]	Biodiversity strengthening	Soil improvement
Water quality improvement	→	↑ *	↘	↗
Soil improvement	↘	↑	↗	
Biodiversity strengthening	↘	↗		
Landscape improvement[b]	↗			

[a] With the perspective of finding the optimal location at catchment scale
[b] With the perspective of water quality improvement at catchment scale
* Landscape improvement referred to landscape heterogeneity

Figure 1. Conflicts between design and management implications in the restoration/construction of wetlands with different perspectives. Vertical arrow means synergy, oblique upwards means compatible with potential limitations, horizontal means no interactions and oblique downwards means compatibility problems.

Landscape approach

Conflicts can also exist when planning wetland restoration at the catchment scale. If soil improvement is pursued in a mosaic scattered throughout the catchment, it could clash with landscape integration recommendations, which advise working in the lower parts of small catchments (<2,000 ha), because of hydrological and economic restrictions related to the distance to permanent streams (Moreno et al., 2008b). A similar situation could appear when restoring with the purpose of strengthening biodiversity, which could also be scattered throughout the catchment. During the process of wetland restoration planning at the catchment scale, restrictions of landscape integration with regard to soil improvement and biodiversity strengthening might be seriously considered, especially when working in small catchments (<2,000 ha). In these cases, the ecological functions of scattered wetlands essential for biodiversity strengthening, such as the facility of colonization or the maintenance of metapopulations of more isolated wetland patches, could be endangered for some species (Foppen et al., 2000).

Recommendations of performing landscape integration in the lower parts of catchments could also present problems with landscape improvement from the perspectives of water quality improvement, where scattered distribution is again advised, and landscape heterogeneity improvement (Fig. 1 and Table 2). However, the landscape integration and landscape improvement perspectives of wetland restoration at the catchment scale also show interesting compatibilities, because both support the recommendation of working in small catchments and the ratios of wetlands area needed for water quality improvement and suitable areas for wetland restoration are compatible. It was pointed out that the highest diversity in agricultural catchments was already higher in lower parts of the catchment (Moreno et al., 2007), which means that an initial effort toward restoration in the proper sense had already been made, and we must simply work to support these natural process. If the catchments area was larger, then the spatial distribution of restored wetlands could be more scattered because the ramification of permanent streams would be more dense, and therefore, there were more areas close enough to the streams to allow for feasible restoration.

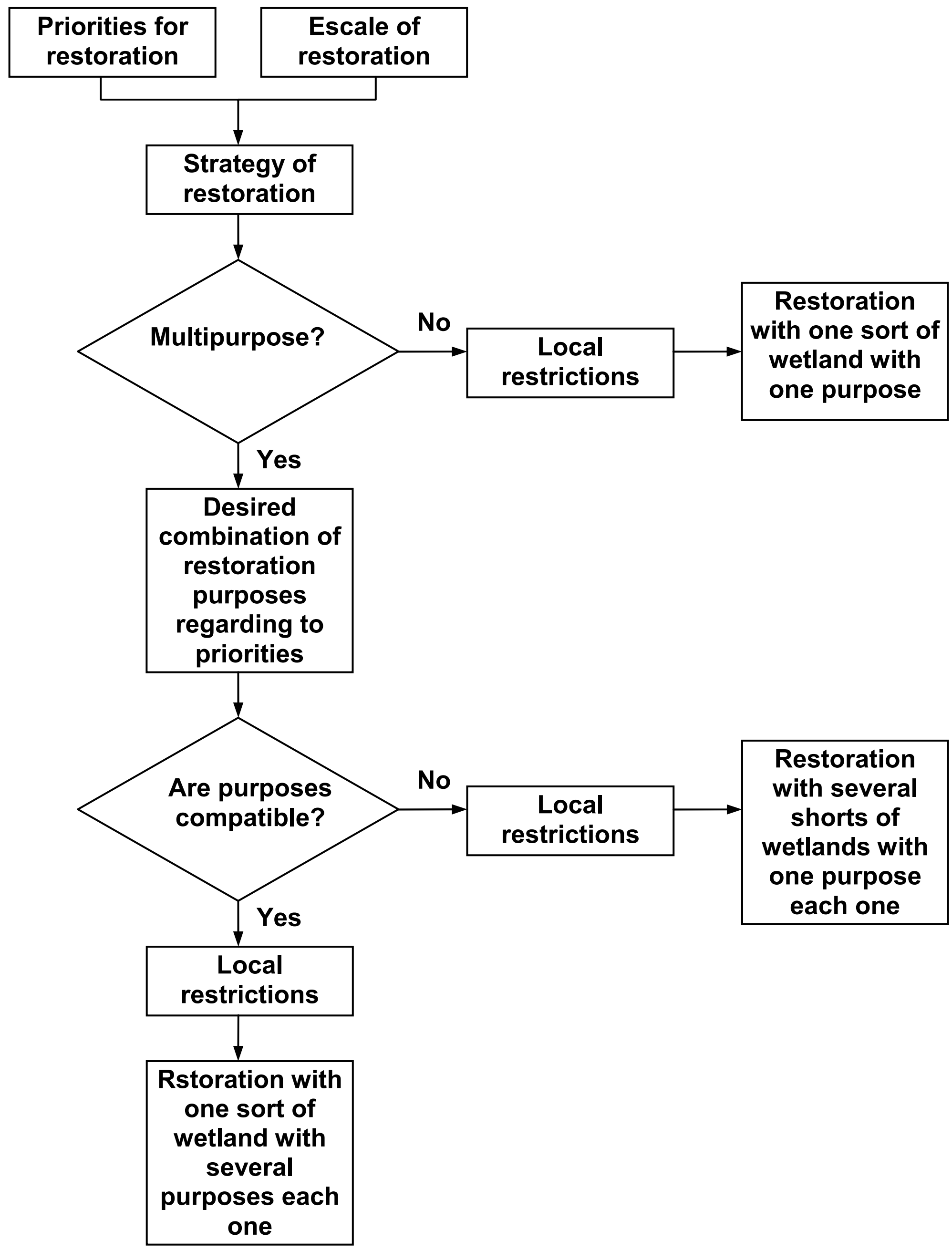

Figure 2. General protocol for planning wetland restoration projects with uni- or multipurpose perspectives.

Other conflicts and compatibilities

Other potential conflicts could occur when plant diversity is searched to strengthen bird diversity, and some areas are maintained only centimetres over the water level. Then, salt ascent from deep layers could be facilitated (Rodríguez-Ochoa et al., 1998), and salinization/solidification could occur in those elevated areas. Finally, a compatibility exists between soil improvement and landscape improvement for water quality improvement;

the higher the water quality, the larger the amount of soil restored because of the amount of wetlands needed to achieve 99% of N removal, up to a limit of ~5% of the catchment area. Equally, the larger the amount of wetlands restored, the higher the landscape diversity of the catchments homogenised by intensive agricultural use.

A protocol for planning wetland restoration

Local limitations

Once we know the local needs for wetlands restoration and the incompatibilities between restoration objectives, we can define a tentative protocol for wetlands restoration in local and regional planning. Initially, we might consider that every restoration place has unique features that will determine the complete restoration process because of its local limitations (soil, topography, hydrology, climate, plant communities, etc.) and the social and ecological needs of the region (water quality, biodiversity, soil improvement or flood control). Local limitations might be studied in depth before initiating any restoration activity. This could be done prior to restoration by compiling existing data or site information (Hopfensperger et al., 2007), or later by, for example, an adaptive approach which continuously revises the current restoration and management activities, proposing alternatives to improve the likelihood of success (Zedler, 2003). The social and ecological needs of the area where restoration will take place must guide the restoration strategy, which will be the result of establishing the objectives and scale of the restoration activities (Fig. 2). To define the strategy, relevant restoration objectives must first be identified and hierarchically ranked, as proposed by Benyamine et al. (2004), according to local or regional priorities. Secondly, we should know the scale at which restoration will be implemented.

Scale of restoration

We propose four typical scales used in prioritizing restoration activities: site, catchment, basin, and territory (Kershner, 1997; Nehlsen, 1997). *Site* refers to one place where one wetland can be restored; *catchment* refers to the area draining surface run-off towards the site (<2,000 ha) where few wetlands can be restored, *basin* refers to river watersheds (hundreds of thousands of hectares) with more than one site/wetland where specific plans usually exist and the territory is a large portion of land with similar climatic, ecological, or social features and can involve several basins or portions of them. Depending on the scale, the implementation of one or another objective of restoration is more or less recommendable (Fig. 3). The larger the scale, the higher the number of objectives that could possibly be compatible, but every restoration objective has an optimal scale to be implemented where the possibilities of success could be higher.

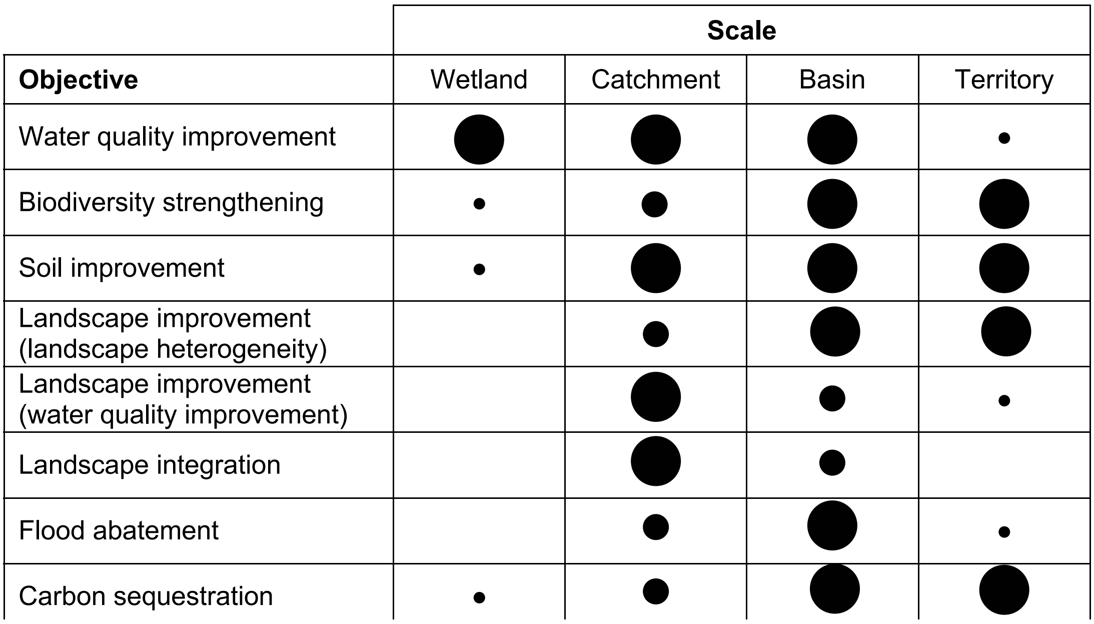

Figure 3. Suitability of restoring wetlands with different purposes according to the scale of implementation (larger dots indicate higher suitability).

At the site scale, only WQI is an advisable objective. The objective could be, for example, to treat the source of pollution (cattle farm or small urban or industrial areas) because all the remaining objectives require a broader perspective in order to be functional. This happens with biodiversity, which is extremely difficult to strengthen with only one wetland. The same happens with soil and all landscape or spatial perspectives.

Increasing the scale at the catchment level, all objectives that need a scattered spatial distribution will increase. Then, we could combine water quality improvement with biodiversity strengthening, soil improvement or landscape improvement (this refers to landscape heterogeneity improvement) if they are compatible with higher probabilities of success (Fig. 4d and e). Biodiversity strengthening and landscape improvement (for landscape heterogeneity improvement) increase their success at the basin scale. Then, the spatial distribution over a wide range will more successfully ensure the establishment of stable bird populations (biodiversity strengthening) and the improvement of the landscape heterogeneity. landscape improvement (for water quality improvement) and landscape integration have higher scores at the catchment scale than at the basin scale because they must be implemented in small catchments (<2,000 ha; Table 2) rather than large basins.

The territory scale could present problems because planning is carried out in multiple basins, and portions of them have different characteristics, and a homogenous restoration plan is not advisable (Fig. 2). It is better to specifically configure restoration projects for each basin, especially when we desire to treat problems related to water quality. Regardless, objectives which require a scattered spatial distribution [biodiversity strengthening, soil improvement or landscape improvement (this refers to landscape heterogeneity improvement)] that are not necessarily located in a specific catchment or basin can be optimally planned at the territory scale.

Figure 5 shows how the maximum suitability for wetland restoration with multiple objectives is achieved at the catchment and basin scales, descending on the territory scale. Zedler (2003) showed that the optimal scale for planning wetlands restoration is at the catchment or basin level for water quality improvement, flood abatement, and biodiversity preservation. We also support the idea that the best scale for wetland restoration planning is at the catchment or basin scales (Fig. 3 and 5).

Strategy of restoration

Zedler (2003) also stated the need for learning how to configure a wetland restoration project to provide the desired mix of ecosystem services at the catchment and basin scales. In our case, knowing the objectives and the scale of restoration, we can configure the restoration project by defining our specific strategy for restoration (Fig. 2). Simple strategies will be those with just one purpose (usually water quality improvement or wildlife preservation) at every scale, but considering local limitations. The potential scenario in this case is shown in figure 4a or c. If several objectives of

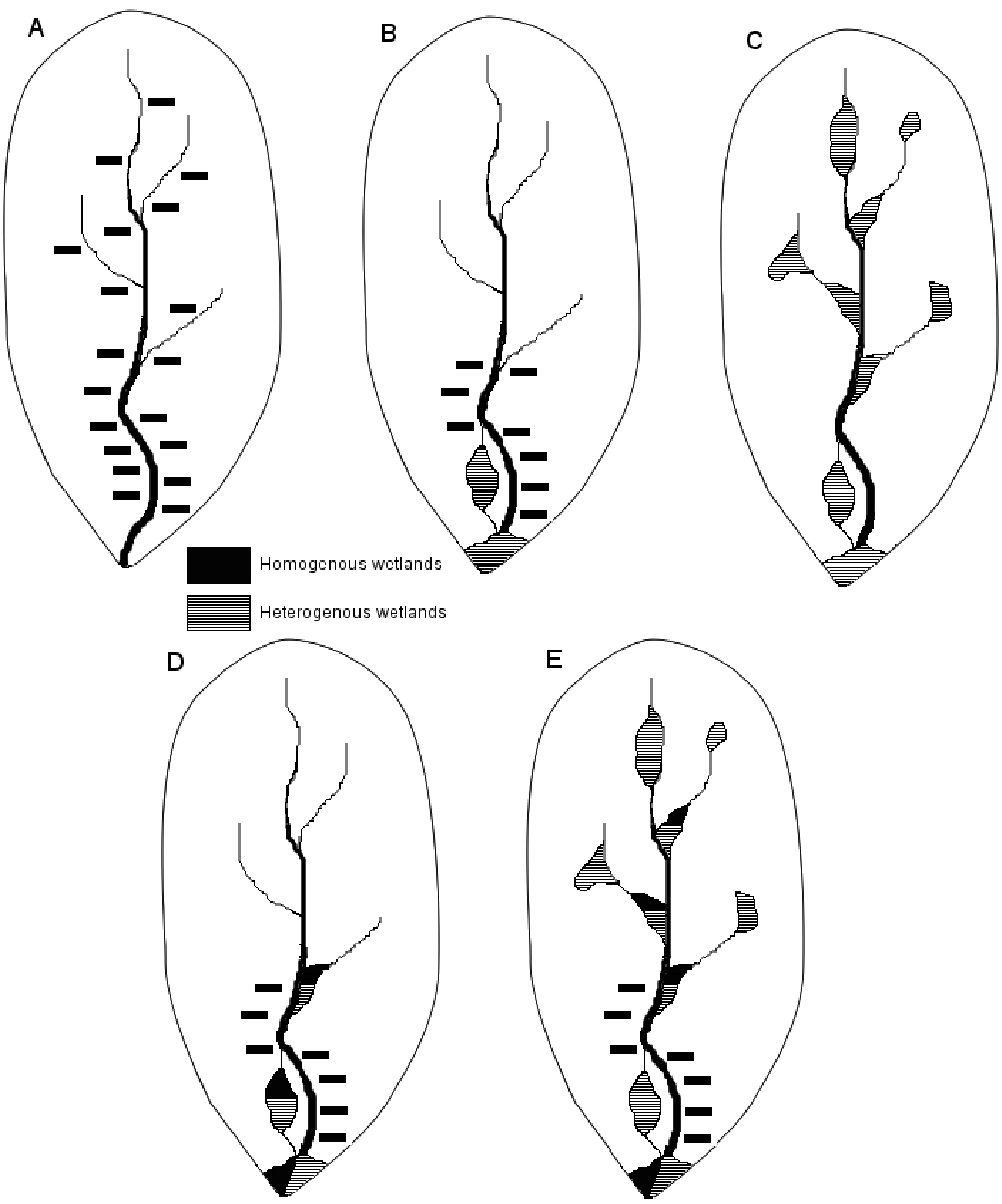

Figure 4. Schematic scenarios for agricultural catchments with different perspectives of wetland restoration or construction. A: Wetlands constructed only with the perspective of water quality improvement, close to 5% of the catchment are wetlands. B: Restoration or construction of wetlands with the perspective of landscape integration within the optimal location (under economic and technical criteria; see Moreno et al. 2008e), water quality improvement and biodiversity strengthening could be combined with the lowest ratio in the catchment (3%). C: Restoration of wetlands only with the perspective of biodiversity strengthening reaching more than 5% of wetlands in the catchment. D: Potential scenario where water quality improvement and biodiversity strengthening are pursued constrained by the perspective of the optimal location (3% of the catchment are wetlands). E: Potential scenario where restoration or construction perspectives of biodiversity strengthening and water quality improvement have no restrictions (maximum ratio of wetlands in the catchment-5%).

restoration are desired, we must find compatibilities and conflicts between them in order to know whether we will restore only one sort of wetland with several objectives, in which case, the scenario would be as in figure 4d without black rectangles, or several sorts of wetlands with different objectives (Fig. 2), where the scenario would be as represented in figure 4b. According to this dilemma, Hansson et al. (2005) stated that such diversified use may be applicable at the catchment scale. We agree with this conclusion, but specify that if catchments are large (with regard to the territory and basins, for example), the likelihood of success is higher for most of the restoration objectives (Fig. 5).

Of course, the restoration success will depend on the combination of efforts dedicated to accomplishing the objectives at different scales, and these are related to the degree of degradation of the different areas/scales under consideration (Lindig-Cisneros and Zedler, 2000).

We should consider that even in restoration projects where only one objective is pursued, more indirect benefits can be naturally obtained. In this sense, Comín et al. (2001) found important benefits for local bird communities and landscape diversity, and Yang et al. (2006) found a general improvement of the entire wetland ecosystem in wetlands both initially designed for water quality improvement. This suggests that even in restoration projects where only one objective is desired, other indirect objectives must be noted, and eventually, supported.

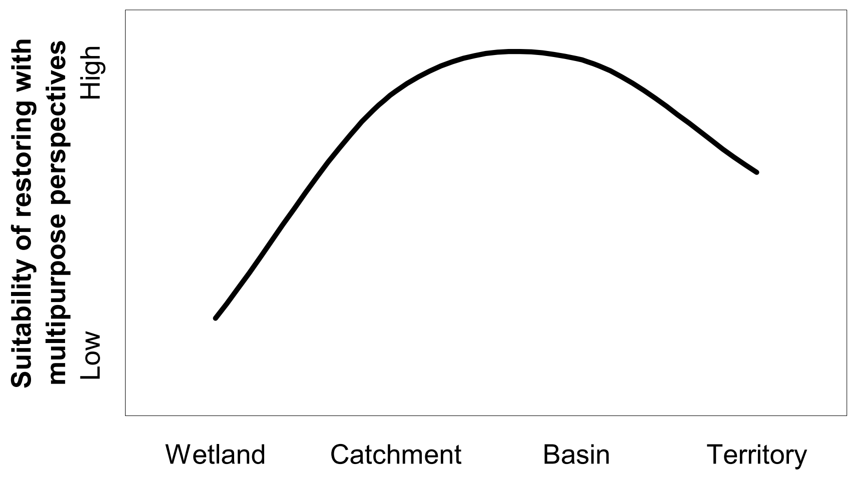

Figure 5. Evolution of restoration suitability of multipurpose wetlands at increasing the scale of landscape planning. The curve was obtained from the weights assigned to each restoration purpose according to its suitability.

Study case: Monegros region

In the particular case of the Monegros region [see detailed description in Moreno et al. (2007)], we suggested three priorities for wetland restoration, hierarchically ranked as water quality, biodiversity, and soil quality. The restoration project was to be planned at the catchment scale. Then, following the proposed protocol (Fig. 2), we defined our restoration strategy. We desired to restore wetlands with multiple functions if possible. Our purposes were water quality improvement, biodiversity strengthening and soil improvement, in that order. water quality improvement is compatible with biodiversity strengthening, nevertheless we would miss large wetlands interesting for more complex bird communities, but this was a secondary priority. We found two local limitations to this approach based the thorough knowledge of the area to be restored. First, the soils presented salinization/sodification problems, and second, streams had permanent water, mainly in the lower parts of the catchments. These limitations showed that wetlands needed to be restored with a permanent flooding, and they could not be scattered all around the catchment. To achieve the first condition, we needed find areas where permanent water is ensured throughout the whole year. The second condition clashed with the biodiversity strengthening and soil improvement objectives, but those were secondary. We also knew that wetlands in the area did not remove P under the current conditions; since water quality improvement was a primary condition, wetlands needed be created to fulfil this objective.

Therefore, the restoration strategy was to construct two types of wetlands. One type was to have the multiple purposes of water quality improvement (to remove N), biodiversity strengthening (with simple bird communities), and SI (in the lower parts of the catchments). Those wetlands needed to be more or less irregularly shaped, with plant diversity and no larger than 5,000 or 6,000 m^2 in size, constructed in lower parts of the catchments close to permanent streams. The second type was to focus on removing P; it could be integrated into the other wetland or into a nearby wetland to treat the same water. This former option needed to follow considerations specific for this objective (P removal), including a deep water table, little vegetation, and removable sediments (Kadlec and Knight, 1996; Hansson et al., 2005). The final design at the catchment scale was similar to that illustrated in figure 4d.

Conclusions

A growing interest exists in restoring wetlands, not only with new approaches (landscape improvement, soil improvement or carbon sequestration) in contrast to the classical approaches (water quality improvement, biodiversity preservation, or flooding control), but also by combining some of these objectives in one or more wetlands in the landscape.

Conflicts and compatibilities between restoration objectives must be carefully studied. A scattered spatial distribution at the catchment scale of restoration planning is crucial in achieving the objectives of biodiversity, water quality and soil improvement which could be carefully combined. Small and homogeneous wetlands are needed for water quality improvement in contrast to the irregular, diverse, and large sites needed for biodiversity strengthening. Problems could appear if plant diversity is attained, leaving areas under intermittent flooding or as islands in the wetland under saline conditions. Improvement of landscape heterogeneity would be more successful working on large catchments (>2,000 ha).

A strategy of restoration must be clearly defined by establishing a hierarchy of priorities based on local needs, and determining the scale of the restoration project. In this sense, we consider that the optimal scale will be at the catchment and, especially, basin scales. A deep knowledge of local limitations must be attained to improve the likelihood of success. Even in restoration projects with one objective at the site scale, secondary objectives could be supported.

Restoration projects using multiple approaches adapted to local needs will provide more services than those with a unique perspective and with a slightly higher investment. Restored and constructed wetlands would be more similar to natural ecosystems and would possess more similar ecological functions, especially if spatial distribution is taken into account during restoration planning. Restored and constructed wetlands could then provide an array of integrated services adapted to local ecological and social needs, mitigating the human impacts on degraded agricultural catchments.

Acknowledgments

This work was supported by CICYT (REN2003-03040), Diputación Provincial de Huesca "Félix de Azara" Awards 2007 and Aragon Government –D.G. Investigación, Innovación y Tecnología– Research Group on Ecological Restoration (E 61).

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Chapter 9

General conclussions/Conclusiones generales

1. La reducción del nitrógeno transportado por el agua sobrante del riego agrícola al pasar por los humedales alcanza el 50% en masa el segundo año de funcionamiento y llega hasta un 99% el cuarto. Este último caso, supone una eliminación anual de 50-90 kg de nitrógeno por hectárea de humedal. Esta retención se alcanzó en humedales con una superficie de 5.000 m^2 y un periodo de retención del agua de unos 4 días.

Al cuarto año de funcionamiento de los humedales se ha alcanzado el óptimo (99%) de reducción de nitrato en el agua excedente de riego. Esto sólo se consiguió en los humedales de 5.000 m^2, en los de menor tamaño (50, 200 y 800 m^2) las reducciones oscilaron entre el 30 y el 50% de la cantidad de nitrógeno total transportada por los canales que recogen el agua sobrante del riego agrícola con similares periodos de retención. Es esperable que el nitrógeno se acumule en el humedal en forma de materia orgánica como consecuencia de su utilización y transformación por la vegetación. Sería interesante en investigaciones futuras comprobar las emisiones de gases de efecto invernadero (metano y óxidos de nitrógeno) que existen en los humedales, para completar un balance y valorar los beneficios y perjuicios que generan estos sistemas de mejora de la calidad de agua.

2. Las características de los suelos sobre los que se asientan los humedales ejercen una influencia importante sobre la calidad del agua que pasa por ellos. Antiguos suelos agrícolas, calcáreos y salinizados restaurados con humedales favorecen la exportación de fósforo, carbono y sales. El incremento de la concentración de sales (fundamentalmente NaCl) en volumen que sufría el agua al

atravesar los humedales experimentales pasó de un 35% el primer año a un 9% el cuarto año de funcionamiento.

Según estos resultados, es previsible que pasados los cuatro años la liberación de sales se irá acercando progresivamente a cero desde el actual 9%. La exportación de fósforo y carbono con el agua que sale de los humedales se podría previsiblemente reducir acortando el periodo de retención actual de unos cuatro días, lo que podría afectar negativamente a la retención de nitrógeno. La comprobación de esta hipótesis y establecer un balance entre el carbono exportado y el transformado en materia orgánica y acumulado en el suelo son aspectos interesantes para futuras investigaciones. Para reducir la emisión de sales es esencial asegurar una inundación permanente que evite su ascenso por capilaridad desde los horizontes más profundos. También es interesante para futuras investigaciones comprobar la evolución de la liberación de sales del suelo con el paso de más de cuatro años.

3. La acumulación de carbono en las capas superficiales del suelo (30 cm superiores) de los humedales existentes en el área de estudio es extremadamente lenta (0.22 mg de carbono por g de suelo y año).

La aridez y salinidad específicas de la zona de estudio no favorecen la acumulación rápida de materia orgánica en el suelo de los humedales y por lo tanto de retención de carbono. En los humedales experimentales no se detectó un incremento significativo de materia orgánica acumulada tras siete años de existencia de la comunidad vegetal. Sería interesante conocer en detalle que factores que limitan la acumulación de materia orgánica y como actúan.

4. La concentración medida en masa en los suelos (30 cm superiores) de nitrato y fósforo disponible fue un 70% y un 35% respectivamente, menor en los suelos de los humedales que en suelos agrícolas de referencia. La concentración medida en masa de las sales (fundamentalmente NaCl) fue un 70% menor en los suelos de los humedales que en suelos de campos agrícolas tras cuatro años de funcionamiento.

Es previsible una rápida eliminación total de los nutrientes inorgánicos del suelo remanentes de la fertilización agrícola convencional que se practicaba en los campos antes de su abandono. El nitrato y el fosfato son de fácil absorción por las plantas. La reducción de sales del suelo implica una liberación a la red hídrica, por lo que la funcionalidad ideal de estos humedales no se alcanzaría antes de que desapareciese esa liberación, lo que ocurre a partir del cuarto año de funcionamiento como humedales.

5. Con el objetivo de fortalecer la diversidad, riqueza y abundancia de las comunidades de aves los humedales deben tener contornos de forma irregular, una alta diversidad de vegetación, un tamaño mínimo de 0,6 ha y deben estar distribuidos de forma dispersa en el paisaje. Además, se deben evitar las quemas incontroladas.

La incorporación de biodiversidad en zonas semiáridas con riego intensivo profundamente transformadas y con paisaje altamente homogeneizado requiere medidas que cumplan eficazmente esta función ocupando una pequeña superficie en el total de la cuenca. Restaurando y creando humedales se puede favorecer el asentamiento de comunidades de aves diversas y abundantes en reducidas áreas del territorio siempre que se haga con el diseño apropiado. Sería de enorme interés estudiar cuales son las características de los humedales que favorecen otros grupos de animales (micromamíferos, reptiles, anfibios, insectos y macroinvertebrados acuáticos) y ver que características benefician o perjudican a cada grupo.

6. Una distribución espacial dispersa y numerosa de los parches del paisaje ocupados por humedales restaurados en la cuenca mejora la calidad del agua más que una distribución espacial agregada y con un reducido número de parches, aunque la superficie total de humedal sea similar.

La complejidad del paisaje, medida como la densidad y agregación de los parches que componen el paisaje y la heterogeneidad (índice de diversidad Shannon para los usos del suelo), favorece la reducción de algunos parámetros de la calidad del agua (concentración de sales y sólidos) de los excedentes agrícolas. También sería importante reducir el tamaño y densidad de los parches del paisaje ocupados por tierras cultivadas en la cuenca, lo que podría conseguirse reduciendo el tamaño de la cuenca sobre la que actuar o aumentado la superficie ocupada por vegetación natural. En este punto se destaca la idoneidad de establecer planes de restauración de humedales, en este caso para la mejora de la calidad del agua, a una escala de cuenca. El siguiente paso de investigación experimental podría ser establecer una cuenca experimental en la que se establecieran el número necesario de humedales y con el diseño apropiado para mejorar la calidad del agua de toda la cuenca.

7. Se ha estimado que es necesario que los humedales distribuidos por las cuencas agrícolas de regadío en zonas semiáridas mediterráneas representen entre el 3,5 y el 6,5% del área de la cuenca para conseguir una eliminación casi total (~99%) del nitrato contenido en las aguas residuales agrícolas.

Aunque la distribución espacial de estos humedales restaurados debería ser dispersa, se ha encontrado que en la mayoría de las cuencas la restauración sólo es posible en los tramos bajos. Esto es debido a que sólo en estos existen cauces de agua permanentes que aseguren un flujo continuo. Esto es importante por los graves problemas de salinización que presentan los suelos con inundación intermitentemente por el ascenso de las sales por capilaridad desde los horizontes más profundos.

8. Las características necesarias para optimizar la retención de nitratos del agua en los humedales (lámina uniforme de agua, vegetación densa y forma compacta) pueden suponen un conflicto con las necesarias para favorecer la diversidad de aves (vegetación heterogénea favorecida por zonas más o menos profundas y formas irregulares).

Consecuentemente, en la planificación de la restauración de humedales a escala territorial se puede plantear la restauración de humedales distintos con diferentes objetivos, o bien, humedales heterogéneos que cumplan los dos objetivos. Además, estas características (forma compacta, vegetación densa y lámina uniforme) deben ser favorecidas por la actividad de restauración, ya que son difíciles de alcanzar de forma natural.

9. Las cuencas hidrográficas pequeñas tienen una mayor diversidad del paisaje que las más grandes.

Esto es debido a la mayor tasa de transformación agrícola (tanto en secano como en regadío) sufrida por las grandes cuencas. En las pequeñas persisten más manchas de vegetación natural, tanto originales como alteradas por el régimen hídrico. Este patrón podría repetirse en cuencas grandes utilizando, por ejemplo, humedales para incrementar su heterogeneidad.

10. La localización de los lugares óptimos para la restauración de humedales con criterios económicos, hidrológicos y ecológicos se debe realizar en cuencas pequeñas (<2.000 ha) con cartografía sencilla, un profundo conocimiento de la zona y el apoyo de expertos.

La cartografía sencilla debe ser aquella que esté al alcance de los gestores del territorio sin que suponga una gran inversión de tiempo o dinero. Se deben hacer estudios para conocer las posibles limitaciones consecuencia del tipo de suelo, la geología, hidrología, propiedad del suelo, regulación de usos del suelo, etc. Las actividades de restauración deben centrarse en las partes bajas de las cuencas donde la permanencia de agua en los cauces es mayor. La cantidad de superficie idónea para las actividades de restauración (15%) está en concordancia con la superficie de humedales requerida para la eliminación de nitratos (3-6%). Sería interesante aplicar esta perspectiva a escalas crecientes de cuencas y comprobar si se mantiene la eficiencia de los humedales extrapolando de forma fractal o no los resultados obtenidos en este estudio al aumentar el área total de la cuenca.

11. La restauración de humedales debe ser llevada a cabo con una estrategia de restauración definida por una jerarquización de los objetivos de restauración y de la escala de planificación, conociendo, además, en profundidad las limitaciones locales.

En el presente estudio se consideró que la escala óptima para implementar una planificación de actividades de restauración debería ser a escala de cuenca, preferiblemente, de grandes cuencas (p.e. cuenca del Flumen) donde múltiples objetivos pueden ser abordados. La compatibilidad entre objetivos debe ser objeto de estudio en la planificación para decidir si es mejor un tipo de humedal con varios objetivos, o bien, varios tipos de humedales con diferentes objetivos.

12. La tesis general planteada con esta investigación "es posible recuperar ecosistemas degradados por el uso agrícola intensivo mediante la utilización de humedales restaurados" se ha demostrado cierta. Como conclusión de todo el trabajo se ha demostrado que esto es posible, se han definido las características que deben tener los humedales para cumplir diferentes objetivos y la estrategia conveniente a seguir para conseguirlo.

Para el futuro sería interesante conocer como se cumple esta hipótesis bajo diferentes escenarios posibles de desarrollo de la sociedad humana desde el ámbito local al global, y que implicaciones tendría para la planificación de la restauración mediante humedales de zonas agrícolas regadas intensamente.