UNIVERSIDAD DE MURCIA

FACULTAD DE BIOLOGÍA

Modelling and establishment of plant indicators of ecological status for semiarid Mediterranean saline wetlands

Establecimiento y modelización de indicadores vegetales del estado ecológico de humedales mediterráneos semiáridos salinos

D. Javier Martínez López

2014

UNIVERSIDAD DE MURCIA

TESIS DOCTORAL

Modelling and establishment of plant indicators of ecological status for semiarid Mediterranean saline wetlands

Establecimiento y modelización de indicadores vegetales del estado ecológico de humedales mediterráneos semiáridos salinos

Autor:
Javier MARTÍNEZ LÓPEZ

Directores:
Dr. Miguel Ángel ESTEVE SELMA
Dra. Julia MARTÍNEZ FERNÁNDEZ
Dra. María Rosario VIDAL-ABARCA GUTIÉRREZ

Memoria presentada para optar al grado de Doctor en el Departamento de Ecología e Hidrología.

2014

Contents

List of Figures

List of Tables

Agradecimientos

Ahora que ya veo la luz al final del túnel, me gustaría agradecer a todos aquellos que me han ayudado, enseñado, apoyado y acompañado durante este proceso. No solamente desde que empecé la tesis, sino desde mucho antes. Ante todo quiero darle las gracias a mis directores, porque sin ellos esto no habría sido posible de ninguna de las maneras. Mirando hacia atrás veo que todo esto ha sido para ellos también una apuesta a largo plazo y quiero darles las gracias por confiar en mí para esta tarea. He aprendido mucho de ellos, de cada uno una cosa. Por supuesto también quiero agardecer al Ministerio por la ayuda FPI que cubrió gran parte del trabajo, incluyendo generosas estancias en centros de investigación del extranjero, sin lo cual no hubiera podido embarcarme en este proyecto. Me siento muy afortunado por las facilidades que he tenido, especialmente ahora en estos tiempo de crisis, en los que muchos de los que empiezan o quieren empezar no tienen tanta suerte. Sin ciencia no hay futuro, y no debería ser un lujo inalcanzable poder investigar en España. Hay mucha gente con ganas de trabajar duro que ya no estamos allí y mucha gente nueva que merece una oportunidad.

Mis compañeros del Departamento de Ecología e Hidrología son y han sido otro gran lujo que he tenido la suerte de tener. Especialmente los doctorandos con los que tanto he compartido y aprendido durante largas e inusitadas horas en el gabinete y en el campo. Quiero darle las gracias especialmente a Paqui Carreño por todo el tiempo que hemos trabajado y aprendido juntos, mano a mano. Sin ella esta tesis no sería la misma. Pero tampoco hubiera sido igual sin Marisa, María Sánchez, Viqui, Rosa, María Luisa y mucha otra gente que estaba desde mis inicios en el departamento como Andrés Mellado, Pedro Abellán, David Sánchez, David Verdiell, Thomas Fichtelman... Y todo el resto de compañeros que prefiero no nombrar porque seguro que me dejo a alguien! Una persona que también ha marcado esta tesis es sin duda Jose Antonio Palazón, quien me ha inculcado desde la carrera el uso del software libre y de herramientas que permitan una ciencia reproducible. Gracias también a mis profesores del máster, especialmente a Paco Alonso, porque me ha servido mucho de lo que aprendí para este trabajo.

En cuanto a mis estancias durante la tesis me gustaría darle las gracias a todo el personal de la Tour Du Valat (Francia), y especialmente a mis compañeros de aventura (María, Jocelyn, Deirdre...). Es un lugar fantástico para los amantes de los humedales. Igualmente quiero darle las gracias especialmente a Siobhan Fennessy, del Kenyon College (Ohio, USA), quien me acogió incluso en su familia y de quien pude aprender mucho sobre humedales durante mi estancia allí. Gracias también a Carles Ibañez por su cálida acogida en el IRTA durante el curso de modelización en julio de 2008.

Parte de esta aventura ha sucedido en Ispra (Italia), por lo que no quiero dejar de mencionar a mis nuevos compañeros aquí, que me han apoyado en esta última etapa: Jon, Beatriz, Fuensanta, Nuria... y especialmente a Gregoire Dubois y Philippe Mayaux por haberme dado la oportunidad de trabajar en el JRC-IES.

Como no, también quiero darles las gracias a todos mis amigos, incluidos los que ya no están, por haberme apoyado incondicionalmente.

Me hace mucha ilusión darle las gracias a mi familia, en especial a mis padres y a mis hermanos. Son unas gracias pequeñitas en comparación con las que se merecen por todo lo demás, no sólo por la tesis, pero este trabajo quiero dedicárselo a ellos, como muestra de mi agradecimiento y amor. Y como no, si alguien me ha acompañado, enseñado, sufrido y ayudado desinteresadamente estos últimos cuatro años, esa es María. Sin ella esta tesis no sería la misma, ni yo tampoco.

Resumen

Introducción general

Los humedales contribuyen de forma muy significativa a la biodiversidad y a la provisión de importantes servicios ecosistémicos, incluyendo alimento, agua, protección frente a avenidas y erosión costera y reducción de la contaminación desde fuentes difusas, entre otros (Okruszko et al, 2011; Jenkins et al, 2010; Zorrilla-Miras et al, 2013). Pese a ello, los humedales permanecen todavía insuficientemente valorados en muchos lugares del mundo, sobre todo en áreas en las que no proporcionan bienes directos de mercado (Maltby and Acreman, 2011; Barbier, 2011), como ocurre en el caso de los humedales de ambientes mediterráneos semiáridos. Esta insuficiente valoración explica que los humedales se encuentren entre los ecosistemas más amenazados por conversión a usos agrícolas, contaminación y sobreexplotación.

En el ámbito mediterráneo una de las tendencias de cambio de uso del territorio más generales es la intensificación agrícola a través de un aumento del regadío (Antrop, 1993; Aranzabal et al, 2008). El incremento del regadío tiene importantes efectos sobre el funcionamiento hidrológico de las cuencas, en particular sobre los flujos de agua y de nutrientes, que a su vez se traducen en cambios sobre distintos subsistemas, incluyendo los humedales. Estos efectos son particularmente importantes en los humedales de ambientes semiáridos, que cuentan con flujos hídricos escasos, elevada salinidad y comunidades biológicas adaptadas a tales condiciones. Las relaciones entre el humedal y su cuenca se siguen ignorando en la gestión de la mayoría de estos ecosistemas (Houlahan and Findlay, 2004), lo que supone un serio problema para su conservación.

Es evidente que para avanzar en el uso sostenible y conservación de los humedales necesitamos herramientas fiables y sencillas de aplicar que permitan evaluar el estado de la cuenca y la manera específica en que afecta a la evolución del humedal de que se trate. Los desarrollos recientes en el ámbito de la teledetección y la modelización hidrológica (Serra et al, 2008; Alexandridis et al, 2008; Ji, 2008) están suponiendo un enorme avance en el análisis de las cuencas y de las relaciones cuenca-humedal. Sin embargo, las metodologías disponibles hasta la fecha pueden ser menos aplicables o útiles en algunas situaciones, particularmente en los humedales menos representativos del caso general, como los de ambientes mediterráneos semiáridos. Por ello se necesitan herramientas que atiendan o sean específicas del contexto ambiental y socioeconómico del humedal de interés. De forma más concreta, se necesitan herramientas capaces de evaluar el efecto de la intensificación agrícola, y de las alteraciones hidrológicas asociadas sobre los humedales mediterráneos, especialmente en ambientes semiáridos como el sureste de España.

Los indicadores ecológicos deberían capturar la complejidad de los ecosistemas manteniendo un nivel de simplicidad que permita una aplicación sencilla y rutinaria, aunque en la práctica se suelen definir y desarrollar orientados a problemas y ecosistemas específicos. En las últimas décadas ha tenido lugar un enorme desarrollo en el ámbito de los indicadores ecológicos para múltiples situaciones, problemas y objetivos, proceso favorecido por la progresiva incorporación de los indicadores al ámbito legislativo y de la planificación y gestión ambiental. Sin embargo, un uso generalizado de los indicadores en tales ámbitos requiere que los mismos se mantengan en número limitado, que sean fácilmente comprensibles y que sean relevantes para la gestión y la toma de decisiones. En el caso del estado ecológico de los humedales de ambientes semiáridos, es evidente que indicadores basados en las características fisico-químicas o bióticas de la lámina de agua no serían de gran utilidad de manera generalizada. Por el contrario, la vegetación puede reflejar bien las condiciones ambientales de estos sistemas (García et al, 2009; Caçador et al, 2013) así como el efecto de posibles perturbaciones (Cronk and Fennessy, 2001; Mitsch and Gosselink, 2007), razón por la que se han desarrollado algunos indicadores de este tipo (véase por ejemplo Miller et al 2006; Johnston et al 2009). No obstante, no se cuenta con indicadores semejantes para el caso de humedales salinos de ambientes mediterráneos.

Un reto añadido a los ya señalados es la necesidad de desarrollar herramientas que faciliten el análisis prospectivo al servicio de la conservación de los humedales y la gestión sostenible de sus cuencas. Los indicadores del estado ecológico de los humedales deben permitir no sólo entender la evolución histórica de estos ecosistemas y realizar diagnósticos acerca de su estado actual y su relación con los usos y alteraciones de la cuenca, sino también analizar los cambios esperables en los humedales bajo distintos escenarios socioeconómicos (como un cambio de uso), ambientales (como el cambio climático) o de medidas de gestión (como manejo de los flujos hídricos) utilizando herramientas de enorme potencial prospectivo como los modelos de simulación. Los avances de las últimas décadas en este campo han generado diversos entornos de modelado. No obstante, permanecen algunos retos pendientes, como la transparencia de tales entornos, la compatibilidad entre modelos y su flexibilidad para adaptarse a distintos contextos (Voinov et al, 2004; Jakeman et al, 2006), de forma que constituyan una herramienta útil para la gestión de humedales y sus cuencas. Aunque se cuenta con algunas aplicaciones de modelización de cuencas basadas en soluciones de entorno software libre y abierto (Zhou et al, 2008), se requiere avanzar más en enfoques integrados basados en este tipo de soluciones (Turner et al, 1995; Argent, 2004; Voinov et al, 2010).

Objetivos

La finalidad última de la presente tesis doctoral es la de desarrollar, aplicar y validar un enfoque metodológico integrado, basado en entornos libres y abiertos, específicamente adaptado para evaluar el estado ecológico de los humedales de ambientes mediterráneos semiáridos, relacionar dicho estado con las alteraciones hidrológicas de la cuenca utilizando técnicas de teledetección y elaborar un modelo dinámico espacial de los humedales que pueda ser utilizado en análisis prospectivos.

De forma más concreta, los objetivos de esta tesis doctoral son los siguientes:

1. Mejorar las técnicas actualmente disponibles de delimitación de cuencas y de análisis de imágenes obtenidas por teledetección y optimizarlas para su aplicación en el estudio y seguimiento de humedales en ambientes mediterráneos semiáridos

2. Determinar la relación a largo plazo entre la composición de especies vegetales de cada humedal y el grado de intensificación agrícola de su cuenca, con el fin de formular un índice de estado ecológico del humedal en relación con las alteraciones hidrológicas generadas por dicha intensificación agrícola.

3. Determinar la viabilidad de formular un índice de estado ecológico basado en comunidades vegetales en lugar de especies individuales y, en consonancia con ello, evaluar la potencialidad de las herramientas de teledetección para la cuantificación y seguimiento del estado ecológico de los humedales en ambientes mediterráneos semiáridos.

4. Elaborar un modelo de simulación espacio-temporal a largo plazo del humedal de Marina del Carmolí utilizando entornos libres y abiertos y verificar los resultados del modelo con información procedente de trabajo de campo y de teledetección.

Área de estudio

La Región de Murcia tiene un clima mediterráneo semiárido con una temperatura media anual de 16 °C y una precipitación media anual de 339 mm (Esteve et al, 2006). Los once humedales objeto de estudio, 7 de ellos costeros y 4 de interior, están catalogados como criptohumedales en el Inventario Regional de Zonas Húmedas de la Región de Murcia (Vidal-Abarca et al, 2003) y son: Boquera de Tabala (CR19), Playa de la Hita (CR20), Marina del Carmolí (CR10), Rasall (H1), Alcanara (CR5), Matalentisco (CR4), Cañada Brusca (CR3), Ajauque (CR14), Derramadores (CR15), Sombrerico (CR21) y Lo Poyo (CR13). Los humedales de la Marina del Carmolí, Lo Poyo y Playa de la Hita se encuentran en la planicie del Campo de Cartagena a orillas de la laguna del Mar Menor, que es la mayor laguna costera presente en el Mar Mediterráneo occidental (Conesa, 1990; Conesa and Jiménez-Cárceles, 2007). La laguna y sus humedales asociados, incluyendo el humedal de las Salinas del Rasall, están incluidos en el catálogo de protección internacional de humedales RAMSAR. El humedal de las salinas del Rasall y su cuenca de drenaje se encuentran, a su vez, dentro del parque regional de Calblanque.

Capítulo 2. Metodologías avanzadas de modelización y teledetección de código abierto para la delimitación y monitoreo de las cuencas hidrológicas de humedales

Para el establecimiento de indicadores de estado ecológico de humedales mediterráneos semiáridos son necesarios estudios a escala de cuenca hidrológica que se centren en las presiones antrópicas que influyen en la dinámica de estos ecosistemas. Nuevos métodos avanzados y reproducibles para el modelado de cuencas y la evaluación de las coberturas del suelo son por tanto herramientas esenciales para el monitoreo y manejo de estos humedales. Sin embargo, pocos estudios integrados proponen metodologías avanzadas de código abierto para el modelado y la evaluación de las cuencas hidrológicas de humedales. En este estudio, se aplicó un conjunto de herramientas de sistemas de información geográfica (SIG), desarrolladas con el fin de delimitar fielmente las cuencas de los humedales de estudio, así como los cambios históricos en sus cobertura de uso del suelo. Las cuencas drenantes a once humedales semiáridas salinos mediterráneos se delimitaron aplicando operaciones de álgebra de mapas sobre el modelo digital de elevaciones en la llanura costera del Campo de Cartagena para mejorar la delimitación de cuencas. Se obtuvieron mapas de cobertura de usos del suelo correspondientes a las cuencas de los humedales en los años 1987 y 2008 por medio de la clasificación supervisada de imágenes Landsat. Un conjunto de cuatro índices espectrales fueron incluidos en el análisis de clasificación usando una combinación de bandas con el fin de mejorar la discriminación de la vegetación, los cuerpos de agua, las infraestructuras y el suelo desnudo. A su vez, se aplicó un procedimiento de clasificación iterativo basado en el algoritmo de máxima verosimilitud y en la selección aleatoria de áreas de entrenamiento, y se incluyó también como capa auxiliar la información contextual basada en la segmentación automática de imágenes de las escenas Landsat. Las cuencas hidrológicas obtenidas oscilaron entre 70 y 17.000 hectáreas y su delimitación se mejoró notablemente en la llanura costera del Campo de Cartagena. La metodología de clasificación de imágenes propuesta mostró una alta precisión, mejorando las técnicas de clasificación estándar. La metodología propuesta se basa en herramientas de código abierto y gratuito, que hacen que sea ampliamente aplicable.

Este capítulo ha sido publicado en *International Journal of Geographical Information Science* (Martínez-López et al, 2013).

Capítulo 3. Índices a escala de humedal y paisaje para evaluar el estado ecológico de humedales mediterráneos semiáridos salinos sometidos a presiones hídricas derivadas de la agricultura

Durante los últimos decenios los humedales salinos semiáridos mediterráneos han sufrido fuertes cambios hidrológicos y biológicos, como consecuencia del aumento de la entrada de agua proveniente de zonas agrícolas adyacentes. Se necesitan indicadores específicos para evaluar el estado de estos ecosistemas únicos en relación a las principales alteraciones hidrológicas a nivel de cuenca. A través del estudio a largo plazo de taxones vegetales seleccionados en un conjunto de humedales representativos de la Región de Murcia (sureste de España), junto con la caracterización de las coberturas de uso del suelo de sus cuencas hidrológicas, se buscaron indicadores vegetales de la condición de dichos humedales, que luego se combinaron en un índice de estado ecológico de humedales. Los porcentajes de las coberturas de interés se ponderaron teniendo en cuenta su cercanía al humedal, así como el tamaño de los humedales. Taxones vegetales perennes característicos de dichos ecosistemas fueron muestreados en 1989 y 2008, y se determinaron los cambios significativos en su frecuencia en cada humedal. Se utilizó un análisis de regresión lineal para relacionar la frecuencia de dichos taxones vegetales en los humedales con la condición hidrológica de sus cuencas de drenaje durante el período de estudio. La frecuencia de *Limonium spp.*, *Arthrocnemum glaucum*, *Phragmites australis*, *Tamarix canariensis* y *Atriplex halimus* mostró relaciones significativas con la condición hidrológica de la cuenca de los humedales. Así, diversos taxones indicadores fueron seleccionados y sus frecuencias fueron combinadas en un índice integrado del estado ecológico de los humedales, que a su vez mostró una fuerte relación con la condición hidrológica de la cuenca.

Este capítulo ha sido publicado en *Ecological Indicators* (Martínez-López et al, 2014a).

Capítulo 4. Teledetección de comunidades vegetales como herramienta para evaluar el estado ecológico de humedales mediterráneos semiáridos salinos en cuencas agrícolas

Los humedales semiáridos salinos mediterráneos son ecosistemas únicos que albergan una gran biodiversidad. En las últimas décadas, la expansión de las tierras de regadío ha provocado un agudo desequilibrio hídrico en las cuencas mediterráneas, causando la degradación de estos ecosistemas. La evaluación de la composición de la vegetación se considera una herramienta importante para la evaluación del estado ecológico de humedales y puede ser estudiada mediante teledetección. Este estudio tiene como objetivo desarrollar un índice de estado ecológico de humedales basado en la composición de sus comunidades vegetales, adecuado para humedales semiáridos salinos mediterráneos, así como probar la aplicabilidad de los sensores remotos multiespectrales para cartografiar las comunidades de plantas de dichos humedales. Las comunidades vegetales características de 12 humedales fueron identificadas por medio de trabajo de campo y análisis multivariante en base a los porcentajes de cobertura de los taxones más característicos. Un índice para evaluar el estado ecológico de los humedales fue desarrollado sobre la base de la relación entre la composición de la comunidades vegetales de los humedales y la condición hidrológica de sus cuencas de drenaje. Las comunidades vegetales de los humedales seleccionados fueron cartografiadas por medio de técnicas de teledetección utilizando el algoritmo 'randomforest' para la clasificación supervisada de imágenes espectrales. Siguiendo esta metodología, la teledetección sirvió como una herramienta para la evaluación del estado ecológico de estos humedales a escala regional.

Este capítulo ha sido publicado en *International Journal of Applied Earth Observation and Geoinformation* (Martínez-López et al, 2014b).

Capítulo 5. Librería y modelo espacio-temporal con R de la respuesta de las comunidades vegetales a las presiones hídricas en un humedal mediterráneo semiárido

Los humedales semiáridos salinos mediterráneos son ecosistemas semi terrestres sometidos anualmente a períodos prolongados de sequía, y que albergan una biota rica, endémica y vulnerable. En las últimas décadas, la expansión de las zonas agrícolas de regadío en las cuencas mediterráneas semiáridas ha provocado alteraciones en los regímenes de agua y nutrientes en los humedales. Dichas alteraciones han afectado a sus comunidades vegetales, resultando en la sustitución de la vegetación halófita característica de estos humedales por taxones más generalistas y oportunistas como los carrizales. Con el objetivo de explicar la distribución espacial de las tres comunidades vegetales más representativas en el humedal de la Marina del Carmolí en respuesta a las presiones hídricas de su cuenca de drenaje, se ha desarrollado una librería de modelización dinámico-espacial utilizando R. Las comunidades vegetales del humedal, así como las zonas agrícolas de regadío en la cuenca fueron cartografiadas mediante teledetección en varias fechas entre 1984 y 2008. Un modelo dinámico fue desarrollado inicialmente usando el software 'Stella' y luego se convirtió a código de R por medio del software 'StellaR'. La dimensión espacial se añadió incluyendo algoritmos de vecindad y flujos espaciales que representan la dispersión de comunidades vegetales. La conversión entre las comunidades vegetales fue inducida por el aumento del regadío en la cuenca, y mediada por parámetros espaciales como la distancia a las ramblas que cruzan el humedal y los valores potenciales de acumulación de flujo dentro del humedal. Los resultados del modelo se asemejan a los datos obtenidos mediante teledetección, que muestran que en 2008 la comunidad vegetal compuesta por estepa salina había perdido la mitad de su superficie original, mientras que las comunidades de saladar y carrizal experimentaron un proceso de expansión importante durante el período de estudio. El modelo desarrollado en este estudio está disponible en internet como una librería de R, que incluye todos los datos de entrada necesarios para ejecutarlo, así como documentación asociada. Tales repositorios de software libre en línea se proponen como herramientas de modelización para investigaciones futuras. La librería del modelo desarrollada en este estudio es una herramienta flexible que se adapta tanto a las necesidades de modelizadores avanzados, como a las de los usuarios menos experimentados.

Este capítulo se encuentra en revisión en la revista *Ecological Modelling*.

Conclusiones generales

1. Los resultados de esta tesis han mostrado cómo el estado ecológico de estos humedales mediterráneos semiáridos salinos se ha visto influenciado negativamente por el incremento de la superficie agrícola de regadío en sus cuencas durante las últimas décadas, lo cual pone de manifiesto la importancia de monitorizar sus cuencas de drenaje para su conservación.

2. Se ha desarrollado una metodología mejorada gratuita y de código abierto para la modelización de cuencas de drenaje de humedales en llanuras costeras, que ha permitido delimitar las cuencas de los humedales con gran fiabilidad, especialmente para humedales extensos.

3. Se ha usado y mejorado una metodología gratuita y de código abierto de clasificación supervisada de imagen que optimiza el procedimiento de generación de mapas históricos de usos y coberturas del suelo con gran fiabilidad, permitiendo así el monitoreo a largo plazo de las presiones que se dan en las cuencas de los humedales.

4. Los resultados indicaron que la frecuencia de taxones vegetales característicos de los humedales estudiados se relacionó significativamente con el incremento en la agricultura de regadío en la cuenca durante el período de estudio de 20 años.

5. El índice basado en la composición y frecuencia de taxones vegetales en estos humedales desarrollado, relacionando las áreas agrícolas de regadío en la cuenca con el estado ecológico de los humedales, se propone como una herramienta útil para la evaluación y monitoreo de los humedales mediante trabajo de campo.

6. El índice de condición hidrológica de cuenca desarrollado, dependiente del área del humedal (WARP), permitió relacionar con éxito el porcentaje de áreas agrícolas de regadío y naturales a escala de cuenca con el estado ecológico de los humedales de estudio, con independencia de su tamaño, y se propone su uso para futuros estudios.

7. El establecimiento de comunidades vegetales en los humedales de estudio por medio de trabajo de campo y análisis de ordenación y clasificación, combinado con su cartografía mediante el uso de sensores remotos aerotransportados, permitió la caracterización espacial remota de los humedales de estudio a dos metros de resolución con alta fiabilidad.

8. Un índice de estado ecológico de humedales basado en la abundancia de comunidades vegetales fue desarrollado y propuesto como herramienta que permite su monitoreo y evaluación mediante técnicas de teledetección. De esta manera se demuestra que la teledetección puede servir para el monitoreo y evaluación del estado ecológico de humedales mediterráneos semiáridos salinos en relación con la condición hidrológica de sus cuencas.

9. El modelo dinámico-espacial desarrollado y testado para la Marina del Carmolí explicó con éxito la expansión de las comunidades de carrizal y saladar en respuesta a las crecientes presiones hídricas provenientes de la cuenca durante el período de estudio, y cómo la abundancia y zonación de las comunidades vegetales responden a parámetros tales como la distancia a las ramblas y la acumulación potencial de flujo en el humedal, que influencian la disponibilidad de agua de las plantas.

10. Los entornos de software libre y gratuitos, tales como R, han permitido el desarrollo de un modelo dinámico-espacial aplicado al estudio de humedales mediterráneos semiáridos salinos. En este sentido, la librería 'spdynmod' se encuentra a disposición pública y puede ser usada y mejorada por la comunidad.

1 Introducción general

1.1. Justificación y antecedentes

Los humedales contribuyen de forma muy significativa a la biodiversidad y a la provisión de importantes servicios ecosistémicos, incluyendo alimento, agua, protección frente a avenidas y erosión costera y reducción de la contaminación desde fuentes difusas, entre otros (Okruszko et al, 2011; Jenkins et al, 2010; Zorrilla-Miras et al, 2013). A nivel global, estudios recientes muestran el papel de los humedales como sumideros netos de carbono, cuya contribución es superior a la de otro tipo de ecosistemas (Mitsch, 2013). Pese a ello, los humedales permanecen todavía insuficientemente valorados en muchos lugares del mundo, sobre todo en áreas en las que no proporcionan bienes directos de mercado (Maltby and Acreman, 2011; Barbier, 2011), como ocurre en el caso de los humedales de ambientes mediterráneos semiáridos. Esta insuficiente valoración explica que los humedales se encuentren entre los ecosistemas más amenazados por conversión a usos agrícolas, contaminación y sobreexplotación. Como ejemplo, se estima que se han perdido ya más de la mitad de los humedales mediterráneos (Weber et al, 2010; Ornat and Morales, 2002).

La necesidad de invertir estas tendencias a la desaparición y degradación de los humedales ha sido reconocida en distintas iniciativas internacionales, incluyendo el Convenio Ramsar (Ramsar Convention Secretariat, 2004) y más recientemente la propia Directiva Marco del Agua (European Commission, 2000), que señala la necesidad de proteger los humedales que dependen directamente de las masas de agua, si bien no establece medidas específicas para ello. Sin embargo, se da la circunstancia añadida de que estos ecosistemas no solo dependen directamente de las masas de agua, sino también, y más específicamente, de su cuenca hidrológica.

En el ámbito mediterráneo una de las tendencias de cambio de uso del territorio más generales es la intensificación agrícola a través de un aumento de la superficie de regadío (Antrop, 1993; Aranzabal et al, 2008). El incremento del regadío tiene importantes efectos sobre el funcionamiento hidrológico de las cuencas, en particular sobre los flujos de agua y nutrientes, que a su vez se traducen en cambios sobre distintos subsistemas, incluyendo los humedales. Estos efectos son particularmente importantes en los humedales de ambientes semiáridos, que cuentan con flujos hídricos superficiales escasos, elevada salinidad y comunidades biológicas adaptadas a tales condiciones. La estrecha relación entre el humedal y su cuenca implica que los factores que gobiernan la dinámica del objeto de conservación o protección – el humedal – dependen de un territorio situado fuera del mismo, a una escala espacial diferente y donde en general no operan las estrategias y acciones de conservación aplicables en el humedal. Las relaciones entre el humedal y su cuenca continúan siendo ignoradas en la gestión de la mayoría de estos ecosistemas (Houlahan and Findlay, 2004), lo que supone un serio problema para su conservación.

¿Cómo se puede abordar esta dificultad desde el ámbito de la investigación para la gestión? Es evidente que para avanzar en el uso sostenible y conservación de los humedales necesitamos herramientas fiables y sencillas de aplicar que permitan evaluar el estado de la cuenca y la manera específica en que afecta a la evolución del humedal de que se trate. Los desarrollos recientes en el ámbito de la teledetección y la modelización hidrológica (Serra et al, 2008; Alexandridis et al, 2008; Ji, 2008) están suponiendo un enorme avance en el análisis de las cuencas y de las relaciones cuenca-humedal. Sin embargo, las metodologías disponibles hasta la fecha pueden ser menos aplicables o útiles en algunas situaciones, particularmente en los humedales menos representativos del caso general, como son los de ambientes mediterráneos semiáridos. Es por ello que se necesitan herramientas que atiendan o sean específicas del contexto ambiental y socioeconómico del tipo de humedal de interés. Por ejemplo, aunque normalmente se señale que la intensificación agrícola reduce los flujos hídricos que sustentan los humedales (véase por ejemplo las revisiones de Zedler and Kercher 2005; Maltby and Acreman 2011; Maltby et al 2013), en ambientes semiáridos la intensificación agrícola de regadío puede actuar en dirección opuesta, aumentando los flujos de agua y nutrientes que llegan al humedal y alterando las condiciones oligotróficas y salinas de los mismos. El hecho de que un mismo proceso (intensificación agrícola) pueda generar efectos opuestos en los humedales subraya la necesidad de enfoques que tengan en cuenta los contextos específicos. De forma más concreta, se necesitan herramientas capaces de evaluar el efecto de la intensificación agrícola, y de las alteraciones hidrológicas asociadas sobre los humedales mediterráneos, especialmente en ambientes semiáridos como el sureste de España.

La conservación de los humedales se enfrenta a otra dificultad añadida debido a su carácter de sistema de transición, lo que implica una relación no trivial entre su estructura (composición de especies, comunidades, diversidad, etc.) y su grado de conservación. Por tanto, evaluar la calidad del estado ecológico de los humedales constituye una tarea en ocasiones compleja y que dista de estar resuelta. Los humedales constituyen sistemas de transición entre los terrestres y los acuáticos. En este sentido, muchos humedales son considerados como aguas de transición en el contexto de la Directiva Marco de Agua (European Commission, 2003; Ferreira et al, 2007) y para los que en consecuencia sería necesario evaluar su estado ecológico y establecer rangos para determinadas variables y parámetros que indiquen los niveles de calidad ecológica en estos ecosistemas.

Aunque con significativo retraso respecto al caso de otro tipo de masas de agua (Basset et al, 2006), la Directiva Marco del Agua ha impulsado el desarrollo de indicadores específicos para aguas de transición (Véase por ejemplo Boix et al 2005; Austoni et al 2007; Muxika et al 2007, 2012; Giordani et al 2009; Lucena-Moya and Pardo 2012; Basset et al 2012), cuyo funcionamiento ecológico invalida la aplicación de los indicadores desarrollados para otros ecosistemas acuáticos, como los ríos o las aguas costeras (Muxika et al, 2005; Dauvin, 2007). No obstante, tales indicadores de aguas de transición no pueden ser utilizados en el caso de humedales de ambientes mediterráneos semiáridos, los cuales pueden considerarse como sistemas semiterrestres situados en un gradiente entre las comunidades estépicas y los ecosistemas plenamente acuáticos (Vidal-Abarca et al, 2003; Castañeda and Herrero, 2008b). Estos humedales presentan un avanzado grado de terrestrialización y una alta estacionalidad, y en el caso de los criptohumedales la lámina de agua puede llegar a no ser visible (Williams, 1999; Innis et al, 2000; Carreño et al, 2008). Así pues, es necesario desarrollar y aplicar indicadores de estado ecológico específicamente adaptados a estos humedales de ambientes semáridos para un efectivo cumplimiento de la Directiva Marco del Agua y como herramientas de alerta temprana (Fancy et al, 2009) para un adecuado seguimiento y conservación de los mismos.

¿Qué indicadores de estado ecológico serían adecuados para la evaluación y seguimiento de los humedales de ambientes mediterráneos semiáridos? Los indicadores ecológicos deberían capturar la complejidad de los ecosistemas manteniendo un nivel de simplicidad que permita una aplicación sencilla y rutinaria, aunque en la práctica se suelen definir y desarrollar orientados a problemas y ecosistemas específicos. Los indicadores ecológicos deberían cumplir criterios como los siguientes (O'Connor and Dewling, 1986): (I) facilidad de medición y aplicación; (II) sensibilidad a los procesos generadores de estrés; (III) presentar un comportamiento predecible y conocido frente a procesos de perturbación antrópica y (IV) presentar una baja variabilidad en la respuesta. En las últimas décadas ha tenido lugar un enorme desarrollo en el ámbito de los indicadores ecológicos para múltiples situaciones, problemas y objetivos, proceso favorecido por la progresiva incorporación de los indicadores al ámbito legislativo y de la planificación y gestión ambiental. Sin embargo, un uso generalizado de los indicadores en tales ámbitos requiere que los mismos se mantengan en número limitado, que sean fácilmente comprensibles y que sean relevantes para la gestión y la toma de decisiones. En el caso del estado ecológico de los humedales de ambientes semiáridos, es evidente que indicadores basados en las características fisico-químicas o bióticas de la lámina de agua no serían de gran utilidad de manera generalizada. Por el contrario, la vegetación puede reflejar bien las condiciones ambientales de estos sistemas (García et al, 2009; Caçador et al, 2013) así como el efecto de posibles perturbaciones (Cronk and Fennessy, 2001; Mitsch and Gosselink, 2007), razón por la que se han desarrollado algunos indicadores de este tipo (véase por ejemplo Miller et al 2006; Johnston et al 2009). No obstante, no se cuenta con indicadores semejantes para el caso de humedales salinos de ambientes mediterráneos.

En muchos humedales el seguimiento con trabajo de campo exige una dedicación muy intensiva y realizada por expertos, lo que inevitablemente encarece mucho dicho seguimiento y con frecuenca en la práctica impide que el mismo se realice con la intensidad, extensión o calidad necesarias. Como alternativa, las metodologías de seguimiento basadas en la teledetección y el análisis espacial representan una ventaja considerable, eliminando de forma significativa las restricciones para el seguimiento frecuente y extensivo de los humedales y sus cuencas. Por ejemplo, se ha utilizado la cartografía basada en la clasificación europea CORINE de usos y coberturas del suelo para desarrollar y aplicar indicadores de estado ecológico de los humedales a distintas escalas espaciales (Weber et al, 2010). Sin embargo, permanecen algunas dificultades metodológicas para optimizar la aplicación de tales herramientas al seguimiento de humedales. Por ejemplo, muchos humedales se localizan en topografías muy llanas, lo que dificulta una delimitación precisa de la cuenca con las herramientas estándar de los módulos hidrológicos SIG (Baker et al, 2006a; Callow et al, 2007). Además, es necesario abordar otros problemas, como las insuficiencias de la clasificación de imagen pixel a pixel (Stuckens et al, 2000; Smith and Fuller, 2001) y de forma más general, los problemas de aplicación de las técnicas de teledetección y clasificación de imagen a estudios históricos de la evolución a largo plazo de los humedales y sus cuencas, dada la menor resolución y calidad de las imágenes más antiguas a la hora de buscar áreas de entrenamiento y validación. A lo anterior se une la dificultad de discriminar la vegetación de los humedales a nivel de especie, dado que su identificación puede ser complicada o conllevar mucho tiempo, lo que puede conducir a errores taxonómicos o a costes desproporcionados. En este sentido, las comunidades vegetales contienen más información ecológica que las especies aisladas y son más fáciles de cartografiar utilizando sensores remotos (O'Connell, 2003; Johnston et al, 2009), lo cual puede ser una vía para solventar esta dificultad y así definir el estado ecológico de los humedales utilizando indicadores basados en las comunidades vegetales en lugar de taxones individuales.

Un reto añadido a los ya señalados es la necesidad de desarrollar herramientas que faciliten el análisis prospectivo al servicio de la conservación de los humedales y la gestión sostenible de sus cuencas. Los indicadores del estado ecológico de los humedales deben permitir no sólo entender la evolución histórica de estos ecosistemas y realizar diagnósticos acerca de su estado actual y su relación con los usos y alteraciones de la cuenca, sino también analizar los cambios esperables en los humedales bajo distintos escenarios socioeconómicos (como un cambio de uso del territorio), ambientales (como el cambio climático) o de medidas de gestión (como manejo de los flujos hídricos) utilizando herramientas de enorme potencial prospectivo como los modelos de simulación. Pese a que la modelización dinámica espacial sigue constituyendo un reto bastante difícil de abordar (Chen et al, 2011), se han desarrollado algunos modelos de simulación espacio-temporales que permiten analizar los cambios de uso del territorio (Wang et al, 2012) y sus efectos sobre los humedales (Jørgensen and Bendoricchio, 2001; Hattermann et al, 2006), lo que ha permitido precisar mejor las medidas de restauración y conservación requeridas por tales humedales. En este sentido, los avances de las últimas décadas en este campo han generado diversos entornos de modelado como el SME (Spatial Modelling Environment; Maxwell and Costanza 1997; Costanza and Voinov 2004; Fitz et al 2011), el SSD (Spatial System Dynamics; Ahmad and Simonovic 2004), MOHID (Braunschweig et al, 2004) o Simile (Muetzelfeldt and Massheder, 2003). No obstante, permanecen algunos retos pendientes como la transparencia de tales entornos y la compatibilidad entre modelos (Voinov et al, 2004; Jakeman et al, 2006).

Finalmente, existe un reto transversal en la aplicación de nuevos desarrollos científicos al ámbito de la gestión: la necesidad de aportar herramientas fácilmente accesibles, gratuitas y suficientemente flexibles como para poder ser adaptadas a los contextos específicos de cada caso. Esta necesidad no está plenamente cubierta con muchas de las soluciones actualmente disponibles, basadas en entornos comerciales y habitualmente bajo licencia, lo que impide su plena accesibilidad. Aunque se cuenta con algunas aplicaciones de modelización de cuencas y humedales basadas en soluciones de entorno libre y abierto (Zhou et al, 2008), se requiere avanzar más en enfoques integrados basados en este tipo de soluciones (Turner et al, 1995; Argent, 2004; Voinov et al, 2010).

En relación a las publicaciones científico-técnicas existentes respecto a las líneas de investigación que han estudiado humedales en la Región de Murcia, algunos ejemplos de trabajos pioneros son los de Barberá et al (1990) en cuanto a su importancia para las aves, así como diversos estudios de tipificación e inventariado de humedales costeros (Robledano-Aymerich et al, 1991a,b, 1992; Ramírez-Díaz et al, 1992). Posteriormente y hasta años recientes se encuentra una gran abundancia de trabajos en referencia a su gestión y conservación (Esteve et al, 1995; Esteve, 2003; Ramírez-Díaz et al, 1995; Gómez et al, 1995; Suárez et al, 1996; Caballero et al, 1996; Vidal-Abarca et al, 2000a,b, 2003; Moreno et al, 2001; Calvo et al, 2003), la dinánimca de nutrientes y su uso en el tratamiento de drenajes y en fitoremediación (Gómez, 1995; Gómez et al, 1997; Vidal-Abarca et al, 1998; Álvarez Rogel et al, 2007b; García-García et al, 2009, 2013; García-García, 2013; González-Alcaraz et al, 2013b), la estructura, dinámica y zonación de la vegetación (Ortiz et al, 1995; Caballero, 1999; Álvarez Rogel et al, 2000, 2001, 2006, 2007a,c; Pardo, 2002; Calvo et al, 2002; González-Alcaraz et al, 2013a), los factores ambientales que influyen en la germinación de especies vegetales (Pujol Fructuoso, 2002; Vicente et al, 2007), el efecto de las alteraciones hidrológicas provenientes de sus cuencas de drenaje (Carreño et al, 2007, 2008; Esteve et al, 2008, 2012; Pardo et al, 2008; Robledano-Aymerich et al, 2010), la diversidad faunística asociada (Millán et al, 2009; García Peiró et al, 2009), así como material didáctico de divulgación (Águila Guillén et al, 2007).

Esta amplia y continuada investigación sobre los humedales de la Región de Murcia ha permitido conocer múltiples aspectos de su estructura y funcionamiento. Sin embargo, quedan abiertas diversas cuestiones, como las planteadas en el contexto de esta tesis, relativas a la búsqueda de métodos de evaluación de su estado ecológico a escala regional, mediante el uso de herramientas de teledetección y modelización.

1.2. Objetivos y estructura de la tesis

La finalidad última de la presente tesis doctoral es la de desarrollar, aplicar y validar un enfoque metodológico integrado, basado en entornos libres y abiertos, específicamente adaptado para evaluar el estado ecológico de los humedales de ambientes mediterráneos semiáridos, relacionar dicho estado con las alteraciones hidrológicas de la cuenca utilizando técnicas de teledetección y elaborar un modelo dinámico espacial de los humedales que pueda ser utilizado en análisis prospectivos.

De forma más concreta, los objetivos de esta tesis doctoral son los siguientes:

1. Mejorar las técnicas actualmente disponibles de delimitación de cuencas y de análisis de imágenes obtenidas por teledetección y optimizarlas para su aplicación en el estudio y seguimiento de humedales en ambientes mediterráneos semiáridos

2. Determinar el efecto a largo plazo del grado de intensificación agrícola de la cuenca en la composición de especies vegetales de cada humedal, con el fin de formular un índice de estado ecológico del humedal en relación con las alteraciones hidrológicas generadas por dicha intensificación agrícola.

3. Determinar la viabilidad de formular un índice de estado ecológico basado en comunidades vegetales en lugar de especies individuales y, en consonancia con ello, evaluar la potencialidad de las herramientas de teledetección para la cuantificación y seguimiento del estado ecológico de los humedales en ambientes mediterráneos semiáridos.

4. Elaborar un modelo de simulación dinámica espacio-temporal a largo plazo del humedal de Marina del Carmolí utilizando entornos libres y abiertos y verificar los resultados del modelo con información procedente de trabajo de campo y de teledetección.

La presente memoria de tesis se compone de una introducción general, en la que se incluye la descripción del área de estudio, cuatro capítulos que responden a cada uno de los objetivos expuestos, las conclusiones generales y la bibliografía. A continuación se presenta brevemente el contenido de cada capítulo.

El capítulo 2 tiene por título: *"Free advanced modeling and remote sensing tecniques for wetland watersheds delineation and monitoring"*. En este capítulo se aborda el primer objetivo y se plantean las siguientes cuestiones: (I) ¿Cómo mejorar los métodos actualmente disponibles para delimitar de forma más precisa la cuenca de los humedales en el caso de topografías especialmente llanas?; (II) ¿cómo mejorar los procedimientos de clasificación de imagen, aumentando la fiabilidad de los resultados y permitiendo su aplicación en el seguimiento a largo plazo, incluyendo estudios históricos? Este capítulo ha sido publicado en *International Journal of Geographical Information Science* (Martínez-López et al, 2013).

El capítulo 3, titulado: *"Wetland and landscape indices for assessing the condition of semiarid Mediterranean saline wetlands under agricultural hydrological pressures"*, aborda el segundo objetivo de la tesis a través de las siguientes cuestiones: (I) ¿Existe relación entre la composición de especies de cada humedal y los cambios a largo plazo (20 años) en los usos y coberturas del suelo de su cuenca?; (II) ¿cómo definir, caracterizar y cuantificar dicha relación?; (III) ¿es posible definir un indicador de estado ecológico de los humedales en relación con las alteraciones hidrológicas derivadas de los cambios de uso de la cuenca? Este capítulo ha sido publicado en *Ecological Indicators* (Martínez-López et al, 2014a).

El capítulo 4 se titula: *"Remote sensing of plant communities as a tool for assessing the condition of semiarid Mediterranean saline wetlands in agricultural catchments"*. Este capítulo aborda el tercer objetivo, en torno a las siguientes cuestiones: (I) ¿Es posible identificar las comunidades vegetales de humedales a lo largo de un gradiente de alteraciones hídricas a escala de cuenca?; (II) ¿es posible aplicar un indicador de estado ecológico de los humedales basado en comunidades en lugar de especies individuales? Y ligada a la anterior, (III) ¿constituye la teledetección una solución eficaz, además de coste-efectiva, para evaluar el estado ecológico de los humedales? Este capítulo ha sido publicado en *International Journal of Applied Earth Observation and Geoinformation* (Martínez-López et al, 2014b).

El capítulo 5 tiene por título: *"A spatio-dynamic R model and library of plant communities responses to hydrological pressures in a semiarid Mediterranean wetland"*. En este capítulo se aborda el cuarto y último objetivo de la tesis, a través de las siguientes cuestiones: (I) ¿Constituyen los entornos de software libre y abierto una alternativa viable para desarrollar e implementar modelos dinámicos espacio-temporales?; (II) ¿es posible aplicar dichos entornos para elaborar un modelo dinámico espacial del humedal de Marina del Carmolí y verificar los resultados a largo plazo de dicho modelo con la información disponible procedente de trabajo de campo y del seguimiento histórico por teledetección? Este capítulo se encuentra en revisión en la revista *Ecological Modelling*.

1.3. Área de estudio

Se ha utilizado como caso de estudio un conjunto de once humedales situados en la Región de Murcia, en el sureste de España (38°45’ - 37°23’ Norte y 0°41’- 2°21’ Oeste). El clima es mediterráneo semiárido con una temperatura media anual de 16 °C y una precipitación media anual de 339 mm (Esteve et al, 2006). La mayoría de los humedales poseen un estatus de protección regional y en algunos casos internacional, por su importancia ecológica y naturalística, ya que incluyen especies y hábitats protegidos, así como especies con distintos estatus de amenaza, y por su funcionalidad ecológica en la dinámica de los flujos de nutrientes. Entre sus valores ambientales se encuentran sus comunidades de vegetación halófila (saladares, juncales y estepa salina), la presencia de aves acuáticas invernantes y nidificantes, aves esteparias, invertebrados acuáticos y terrestres, así como de fauna piscícola endémica. Su interés paisajístico y los diversos usos tradicionales que en ellos se han ido realizando son también valores que justifican su conservación.

Los once humedales objeto de estudio, 7 de ellos costeros y 4 de interior, están catalogados como criptohumedales en el Inventario Regional de Zonas Húmedas de la Región de Murcia (Vidal-Abarca et al, 2003). Son humedales carentes de lámina de agua libre en la mayor parte de su superficie, que se desarrollan sobre zonas llanas o de escasa pendiente y drenaje difuso, receptoras de escorrentías y descargas laterales y subterráneas, y son concretamente: Boquera de Tabala (CR19), Playa de la Hita (CR20), Marina del Carmolí (CR10), Rasall (H1), Alcanara (CR5), Matalentisco (CR4), Cañada Brusca (CR3), Ajauque (CR14), Derramadores (CR15), Sombrerico (CR21) y Lo Poyo (CR13) (figura 1.1). Los humedales de la Marina del Carmolí, Lo Poyo y Playa de la Hita se encuentran en la planicie del Campo de Cartagena a orillas de la laguna del Mar Menor, que es la mayor laguna costera presente en el Mar Mediterráneo occidental (Conesa, 1990; Conesa and Jiménez-Cárceles, 2007). La laguna y sus humedales asociados, incluyendo el humedal de las Salinas del Rasall, están incluidos en el catálogo de protección internacional de humedales RAMSAR (Ramsar Convention Secretariat, 2004). El humedal de las salinas del Rasall y su cuenca de drenaje se encuentran, a su vez, dentro del parque regional de Calblanque. La mayoría de sus cuencas de drenaje han experimentado en las últimas décadas diversos grados de alteración hidrológica derivada de la intensificación agrícola, específicamente debido al incremento del regadío.

Las comunidades vegetales características de estos criptohumedales son la estepa salina y el saladar. La estepa salina está compuesta mayoritariamente por el hábitat de interés prioritario *1510 – Estepas salinas mediterráneas (Limonietalia)*, según la Directiva Hábitats. Las principales especies de la estepa salina son *Lygeum spartum*, *Suaeda vera*, *Frankenia corymbosa* y *Limonium caesium*. El saladar está dominado por los hábitats *1420 – Matorrales halófilos mediterráneos y termoatlánticos (Sarcocornetea fruticosi)*, *1430 – Matorrales halonitrófilos (Pegano-Salsoletea)* y *1410 – Pastizales salinos mediterráneos (Juncetalia maritimi)*, siendo sus principales especies *Sarcocornia fruticosa*, *Arthrocnemum macrostachyum*, *Halimione portulacoides* y *Halocnemum strobilaceum*. El hábitat *92D0 – Galerías y matorrales ribereños termomediterráneos (Nerio-Tamaricetea y Flueggeion tinctoriae)* también se da en estos humedales compuesto por *Tamarix canariensis* y *Tamarix boveana*. En la mayoría de estos criptohumedales aparece, a su vez, otra formación vegetal, el carrizal dominado por la especie *Phragmites australis*. La distribución espacial de las unidades de vegetación depende de la disponibilidad del agua, del tipo de sustrato y de las condiciones de salinidad. Todos los hábitats, a excepción del carrizal, son de interés comunitario, siendo la estepa salina además prioritario.

Figura 1.1: Localización de los humedales de estudio en la Región de Murcia (SE España). Leyenda: **H1**: Rasall; **CR3**: Cañada Brusca; **CR10**: Carmoli; **CR5**: Alcanara; **CR14**: Ajauque; **CR13**: Lopoyo; **CR15**: Derramadores; **CR19**: Boquera de Tabala; **CR20**: Playa de la Hita; **CR4**: Matalentisco; **CR21**: Sombrerico.

2 Free advanced modeling and remote sensing techniques for wetland watersheds delineation and monitoring

Abstract

Watershed scale studies focusing on hydrological pressures influencing freshwater ecosystem dynamics are necessary for the establishment of suitable wetland ecological indicators. Enhanced and reproducible methods for watershed modeling and land cover assessment are thus essential tools for wetland monitoring and management. However, few integrated studies propose advanced open source methodologies for watershed modeling and assessment. In this study, a set of GIS methodological tools was applied and further developed in order to delineate wetland watersheds and map their land cover changes over time. Watersheds draining to eleven semiarid Mediterranean saline wetlands were delimited and map algebra operations were applied on the digital elevation model in the Campo de Cartagena coastal plain to enhance watershed delimitation. Land use/land cover maps of wetland watersheds were obtained for years 1987 and 2008 by means of supervised classification of Landsat images. A set of four spectral indices were included in the classification analysis using a combination of bands in order to improve the discrimination of vegetation, water bodies, infrastructure and bare soil. An iterative classification procedure based on maximum likelihood and random selection of training areas was applied. Contextual information based on automatic image segmentation of Landsat scenes was also included as ancillary layers. Watershed areas obtained ranged from 70 to 17,000 ha and delineation was improved in the Campo de Cartagena coastal plain. The proposed image classification methodology showed high accuracies and improved standard classification techniques. The proposed methodology is based on free and open source tools, which makes it broadly applicable.

2.1. Introduction

Wetlands naturally act as a sink of upland occurring drainage (Ji, 2008). Land use changes lead to hydrological alterations at the watershed level, which directly influence biotic communities in freshwater and terrestrial ecosystems (Tong and Chen, 2002; Carreño et al, 2008). The expansion of agricultural irrigated areas and urban and tourist development in the Mediterranean Region, and more specifically in Murcia province (SE Spain), during recent decades has led to relevant hydrological changes that affect watersheds and their associated wetland ecosystems (Esteve et al, 2008). Long term monitoring of land uses/cover at watershed scale is therefore highly important for the proper establishment of indicators of wetland condition (Roth et al, 1996; McHugh et al, 2007). However, few integrated studies propose advanced free open source methodologies for wetland watershed modeling and assessment (Aspinall and Pearson, 2000; Hollenhorst et al, 2007; Zhou et al, 2008).

Delineating specific watershed areas draining to each wetland with high accuracy is essential for the establishment of landscape-wetland relationships, especially in relation to landscape hydrological processes (Felicísimo, 1994; Hollis and Thompson, 1998; Turner et al, 2003). However, standard GIS hydrological modeling modules are not always suitable for proper watershed delineation (Baker et al, 2006b; Callow et al, 2007). Watersheds are usually delineated by the area upstream from a given outlet point. Therefore, when several stream network channels drain into a wetland, their respective outlet points within the wetland area must be accurately located in proper relationship to streams and the flow direction map. Especially in flat areas, identifying relevant outlet points inside the wetland area can be a very time-consuming process, particularly for large wetlands. Enhanced watershed delineation methods must be applied in coastal plain areas, where the drainage network might not be clearly defined in a DEM, in order to obtain accurate results.

Several methodological issues arise when performing land use/land cover mapping by means of supervised classification methods (Lu and Weng, 2007; Horning et al, 2010). Distinctiveness of spectral signatures for each land use/land cover class depends on the available remote sensor bands. However, available bands can be also combined into several indices, which enhance the discrimination of land cover classes (Zha et al, 2003). Standard available classification methods do not account for neighboring pixels. However, landscapes consist of several land cover class patches, often bigger than image pixel size. Pixel based classification methods are highly dependent on pixel size and usually lead to a large amount of isolated pixels of different land cover classes in the resulting map (Stuckens et al, 2000; Smith and Fuller, 2001). Landscape patches of the same land cover class tend to have similar shapes, *e.g.* urban and agricultural areas tend to be geometric whereas natural areas are irregular. Recent studies have applied image segmentation for similar applications, such as forest cover mapping (Magnussen et al, 2004; Hay et al, 2005; Hirata and Takahashi, 2011). Thus, contextual information is needed to enhance pixel based classification methods when generating landscape maps (Yan et al, 2006; Blaschke, 2010; Vieira et al, 2012). The selection of reliable and representative training site sets is also crucial for the proper elaboration of spectral signatures, thus determining classification accuracy. This is especially important for land cover time series maps, for which training sites must be extracted from different sources, often of unequal quality. Since older aerial photographs have lower resolution, reliable training sites are more difficult to obtain.

Free and open source software (FOSS) GIS provides advanced environmental modeling and management tools and is becoming widely used in terms of number of projects, end users and financial support (Steiniger and Bocher, 2009). This study offers an enhanced procedure for the monitoring of watershed pressures on wetlands by means of a better delineation of specific watershed areas and the accurate assessment of their land use changes by means of FOSS.

The objective of this study was to develop and apply a set of enhanced methods to delineate eleven semiarid Mediterranean saline wetland watersheds in Murcia province and to map their land uses in 1987 and 2008 by combining hydrological modeling and remote sensing techniques using free and open source software. Our specific objectives were: (1) to accurately establish wetland watersheds, improving delineation methods in plain areas and for large wetlands; (2) to map land uses in wetland watersheds in years 1987 and 2008 by means of an improved procedure of supervised image classification that (a) uses several spectral indices to enhance discrimination of land cover classes, (b) includes patch scale information in image classification and (c) minimizes the effect of training site intraclass heterogeneity.

2.2. Materials and methods

2.2.1. Study area

Murcia province (SE Spain: 37°N, 1°W) has a semiarid Mediterranean climate with a mean annual temperature of 16 °C and a mean annual precipitation of 339 mm (Esteve et al, 2006). Eleven semiarid saline wetlands were selected, *i.e.* 7 coastal and 4 inland wetlands (Fig. 2.1). Selected sites are included in the regional inventory of wetlands (Vidal-Abarca et al, 2003) and their protection status ranges from regional, national to international level due to their high ecological values (Ramsar Site, Special Protection Area for Birds, Site of Community Importance and Special Protection Area for the Mediterranean), except for Matalentisco and Boquera de Tabala wetlands, which do not benefit from any protection status. Marina del Carmolí, Lo Poyo and Playa de la Hita wetlands are in a lowland coastal plain, called Campo de Cartagena, associated with the internal shore of the Mar Menor coastal lagoon which comprises 12,700 ha (Conesa and Jiménez-Cárceles, 2007). The lagoon and its associated wetlands are all RAMSAR sites, containing eighteen Habitats of Community Interest according to the European Habitat Directive (Council of Europe, 1992). Salinas del Rasall is a coastal wetland associated with a salt extraction pond embedded in the Calblanque Natural Park, and also included in the Mar Menor RAMSAR protected area. Matalentisco, Sombrerico and Cañada Brusca are coastal wetlands located in the southern part of the region on the Mediterranean Sea. Boquera de Tabala, Ajauque, Derramadores and Alcanara are inland wetlands associated with an ephemeral river and with a saline alluvial plain, respectively.

Figure 2.1: Wetlands location map in Murcia province (SE Spain). Wetland keys: **H1**: Rasall; **CR3**: Cañada Brusca; **CR10**: Carmoli; **CR5**: Alcanara; **CR14**: Ajauque; **CR13**: Lopoyo; **CR15**: Derramadores; **CR19**: Boquera de Tabala; **CR20**: Playa de la Hita; **CR4**: Matalentisco; **CR21**: Sombrerico.

2.2.2. Digital elevation model preprocessing and delimitation of wetland watersheds

A 10 m resolution digital elevation model (DEM) of Murcia province created by the Instituto Geográfico Nacional was used for hydrological modeling. Map layers and GIS analyses have been processed with *GRASS GIS* 6.4 software (GRASS Development Team, 2008; Neteler et al, 2012). Map algebra operations were applied on the DEM in the Campo de Cartagena coastal plain to reinforce the existing drainage network and to force flow accumulation from all draining areas around wetland perimeter to converge into a single point within the wetland area. A pixel in an arbitrary coordinate within each wetland area was first selected, which ultimately served as a sink or outlet point, from which to delineate the wetland watershed. Then, a wetland map was created in which remaining pixels inside wetland area were assigned a distance value related to the selected pixel. From this map, a new map was created calculating the inverse distances, showing the highest value in the pixel selected as a sink. This map was then subtracted from the raw 10 m resolution DEM of the Campo de Cartagena. Therefore, a single point was obtained within the wetland area from which to perform watershed delineation in order to collect all drainage fluxes reaching the wetland perimeter. Outside of wetland areas the DEM was also modified by lowering the elevation values coinciding with the existing ephemeral stream network to force flow-direction models to match existing stream lines (Fig. 2.2; Olaya Ferrero 2004; King et al 2005). This is a so-called 'stream burning' operation. The drainage network in the Campo de Cartagena coastal plain was thus forced to match the existing stream network. For the delineation of the remaining wetland watersheds outside the Campo de Cartagena area the raw DEM was used.

Maps of flow accumulation and drainage direction based on the DEM were generated using the single flow direction (D8 algorithm). For wetlands in the Campo de Cartagena area two main inflows coming from the watersheds were obtained, which converged into the previously selected sink point (Fig. 2.3). Watersheds were finally delimited from the selected sink coordinates within the wetlands using the drainage direction maps.

2.2.3. Remote sensing of watershed land use/land cover

The Landsat sensor was selected as a suitable image data source due to its medium spatial resolution and high spectral and temporal resolution provided that: (1) medium spatial resolution is appropriate for regional scale studies; (2) high spectral resolution enhances the discrimination of land cover types; and (3) high temporal resolution allowed us to add phenological information to the classification analysis in order to better discriminate some land cover classes by using two images for each study year. Map layers and GIS and statistical analyses in this study have been processed with *GRASS GIS* 6.4 software and *R* (R Core Team, 2013).

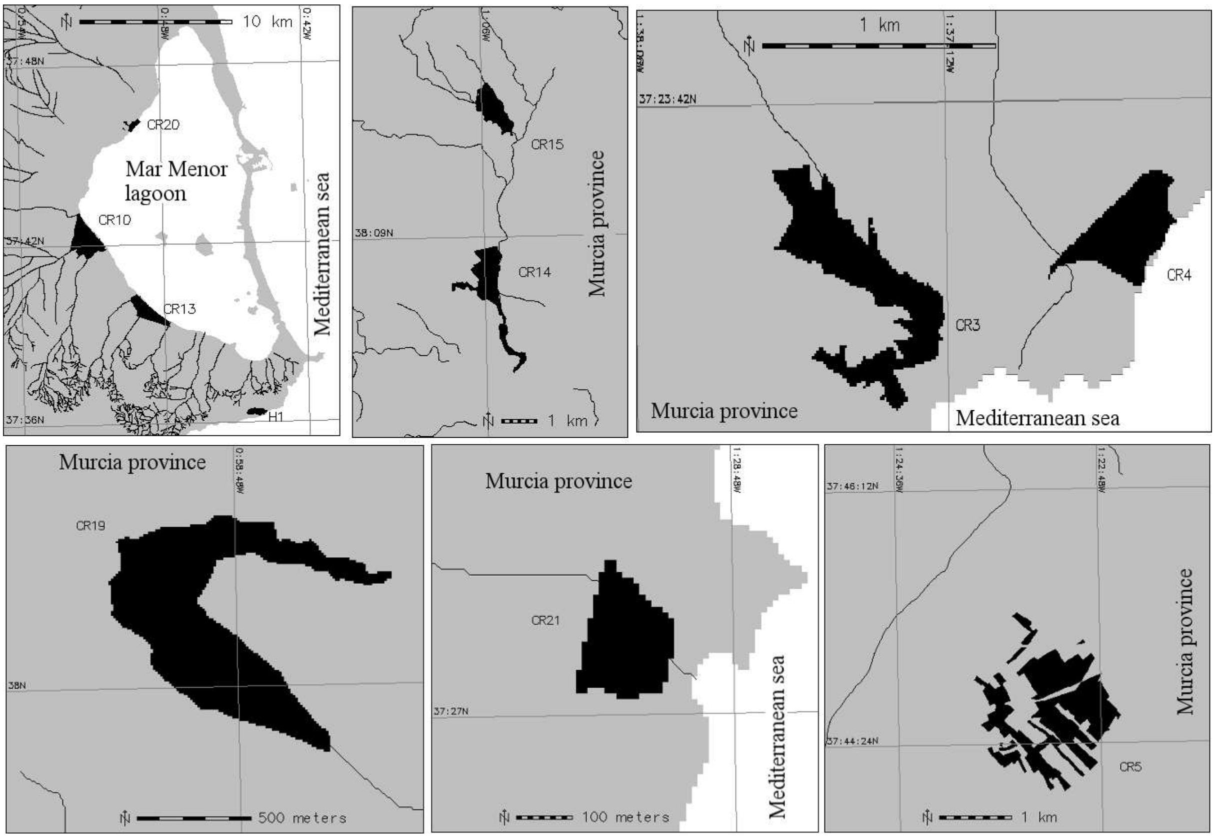

Figure 2.2: Wetlands maps in relation to the ephemeral river network. Wetland keys: **H1**: Rasall; **CR3**: Cañada Brusca; **CR10**: Carmoli; **CR5**: Alcanara; **CR14**: Ajauque; **CR13**: Lopoyo; **CR15**: Derramadores; **CR19**: Boquera de Tabala; **CR20**: Playa de la Hita; **CR4**: Matalentisco; **CR21**: Sombrerico.

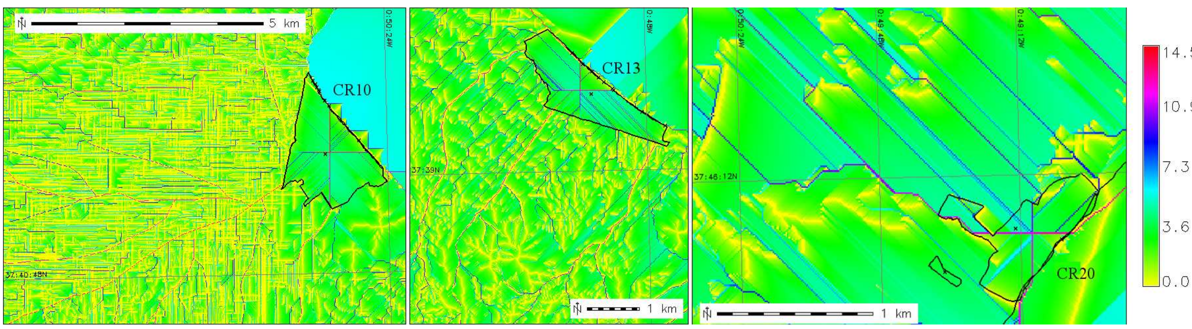

Figure 2.3: Detail of resulting maps of flow accumulation (log scaled) at the Campo the Cartagena wetlands. Left: Marina del Carmolí wetland (CR10); center: Lopoyo wetland (CR13); right: Playa de la Hita wetland (CR20).

Landsat 4 images were used for the classification in 1987, corresponding to the TM sensor, while the ETM+ sensor was used (Landsat 7) for the 2008 classification. Spectral bands used were: blue (B), green (G), red (R), near infrared (NIR) and short-wave infrared (MIR). Pixel size was set to 25 m. Two Landsat images corresponding to different dates, winter and late spring, were used for each classification in order to account for the changing seasonal phenology of vegetation during the dry and wet periods (Bradley and Mustard, 2005). Landsat images from 1987 and 2008 were obtained from the Instituto Universitario del Agua y del Medio Ambiente and from the Instituto Geográfico Nacional, respectively. Atmospheric correction of images was performed using the Dark object subtraction technique (Pat S Chavez, 1988; Chavez, 1996; Chander et al, 2009). Eleven land cover classes were mapped in years 1987 and 2008: (1) dense (DNW) and (2) open (ONW) natural woodland, (3) dense (DNS) and (4) open (ONS) natural shrubland, (5) dry arboreal (DAC) and (6) herbaceous (DHC) cropland, (7) irrigated arboreal (IAC) and (8) herbaceous (IHC) cropland, (9) greenhouses (GHs), (10) unproductive land and infrastructure (INF) and (11) water bodies (WBs).

2.2.3.1 Spectral indices

Four spectral indices for each Landsat image were calculated and were included in the classification analysis as ancillary layers. These indices enhance the discrimination of vegetation, urban areas, bare soil and water bodies. The Normalised Differential Vegetation Index (NDVI) was calculated (Rouse et al, 1973). NDVI is widely used in remote sensing studies for the identification of vegetation since it highlights photosynthetic activity (Bannari et al, 1995). Dense vegetation shows NDVI values closed to one, soil values are positive but lower, and water values are negative due to its strong absorption at NIR (Glenn et al, 2008). The MNDWI (Modified Normalized Difference Water Index; Hui et al, 2008) was calculated to delineate water bodies and enhance its presence. This index can also remove shadow effects on water, which are otherwise difficult to detect. The NDBI (Normalized Difference Built-up Index; Zha et al, 2003) was calculated to enhance the discrimination of built-up areas. Finally, we calculated the NDBaI (Normalized Difference Bareness Index; Chen et al, 2006). This index retrieves bare soil from the Landsat images. Bare soil (including beaches, bare soil, and land under development) could be distinguished by NDBaI values greater than zero.

2.2.3.2 Contextual information

Landscape patches in the study area were extracted by means of automatic segmentation performed on a single Landsat scene for each year using *SPRING 5.1.5* software (Cāmara et al, 1996; see figure 2.4). *Region Growing* data segmentation technique was performed on image bands number 1 to 7 (excluding the thermal infrared band). Similarity threshold (ranging from 0 to 100%) and minimum patch size (in image pixels units) parameters were set to 20 and 100 respectively, following experimentation. Two shape indices for each obtained patch were then calculated and scores were assigned to pixels belonging to each patch. The new maps with each corresponding shape index values were included in the classification as ancillary layers,

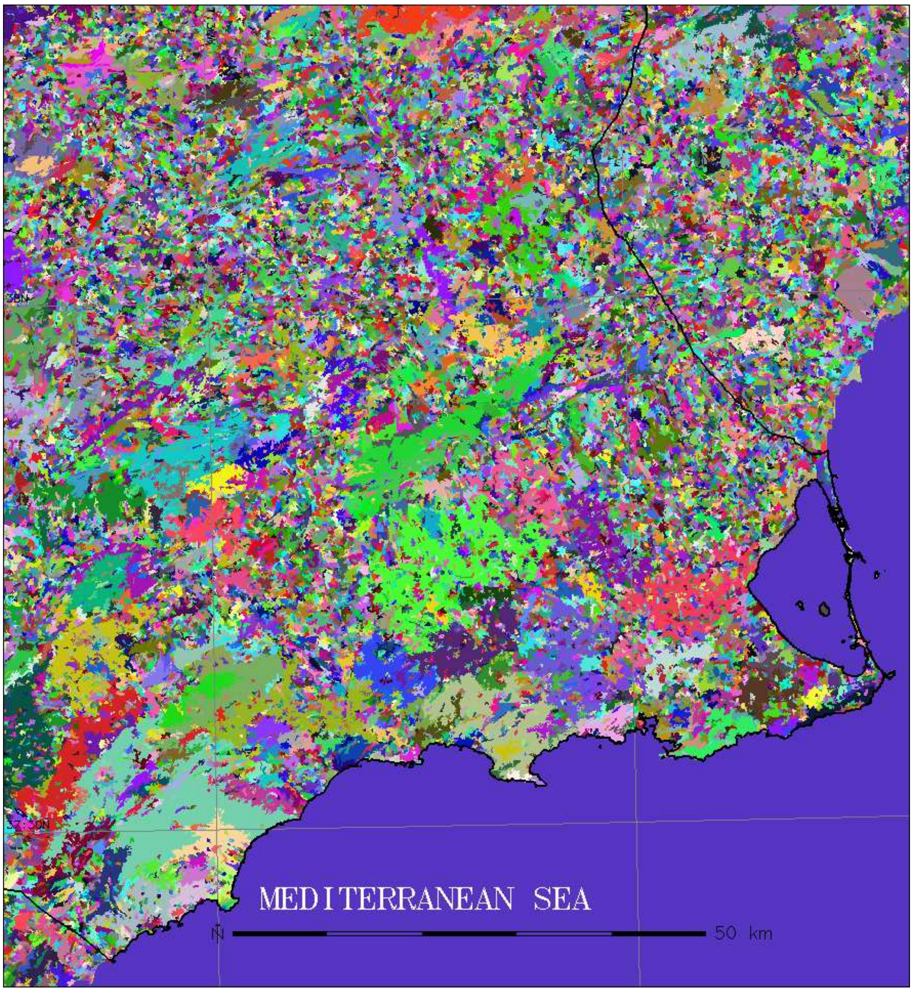

Figure 2.4: Automated landscape patches map obtained with *SPRING* software. Different colors represent different landscape patches.

hence including patch information at pixel scale. Patches from the resulting map were scored according to the shape indices. Two new maps were obtained based on Fractal Dimension (FD) and Shape Index (SI) of patches sensu Chust and Ducrot (2004), which were used as ancillary input layers for classification.

2.2.3.3 Supervised classification: the 'randomclasiter' method

Land use/land cover maps were obtained by supervised classification using the maximum likelihood algorithm (Michelson et al, 2000; Richards and Jia, 2006). Input data for the classification analysis were the resulting shape and spectral indices maps, together with the single bands of the two Landsat scenes except for the thermal infrared band, which was not included in the analysis. Training and validation sites were obtained through aerial photograph interpretation. Classification was enhanced with an iterative procedure that minimizes the effect of uncertainty and heterogeneity in the selection of training sites, called 'randomclasiter'. This procedure was developed by Carreño et al (2011) and the code is available on-line (https://github.com/Paquicf/randomclasiter). The procedure combines *GRASS*, *R* and bash scripts to automatically perform several supervised classifications, randomly selecting a different subset of training areas each time. For the 2008 classification, a total of 521 training areas were selected from an aerial image of Murcia Region from year 2008 (DGPNB, 2008) using QGIS software (Development Team, 2009). For the 1987 classification, a total of 407 training areas were selected from an aerial image of Murcia Region from year 1987 from the Confederación Hidrográfica del Segura. One hundred iterations were performed and for each of them 25-50% of the training areas were randomly selected. This percentage was the same across iterations but different for each land cover class, depending on the total number of training sites available.

In each classification a pixel might be assigned to different land cover classes, depending on the set of training areas selected. One hundred maps were therefore generated and later summarized into eleven maps – one for each land cover class, each of them representing the total frequency of assignment of each pixel to a specific land cover class, ranging from 0 to 100 (*i.e.* the number of iterations). The final land cover class assigned to a pixel in the resulting map was ultimately the class which was most often assigned to this pixel.

The methodology was verified by aerial image validation and stratified random sampling by land cover class. The number of pixels used to validate each land cover class was proportional to the number of training sites. A total of 696 pixels were used as validation areas for year 2008 and 636 pixels for year 1987. Classification results were validated by means of the *Overall Accuracy* parameter and the *Kappa* coefficient (Chuvieco, 2002; Foody, 2002). User's and producer's accuracy parameters were also calculated for each land cover class of interest (Congalton, 1991).

2.3. Results

2.3.1. Wetland watershed delimitation enhancements

Watershed areas obtained ranged from 70 to 17,000 ha (see table 2.1). In the Campo de Cartagena coastal plain area derived watersheds were consistent with previous hydrological studies and were larger than the ones obtained without DEM preprocessing (Fig. 2.5; Conesa, 1990). Results showed that in Cañada Brusca wetland two different wetland areas were drained by distinct watersheds, thus presenting different hydrological influences at watershed scale. Therefore, the wetland was divided into two different sites: Cañada Brusca North (CR3N) and South (CR3S).

2.3.2. Image classification enhancements

Overall accuracy percentages and kappa values for the 1987 and 2008 classifications were higher using the 'randomclasiter' methodology in relation to the standard method (table 2.2), showing higher values in 2008 than in 1987. The inclusion of a set of spectral indices as ancillary layers clearly improved the proper classification of pixels (Fig. 2.6B) in relation to using only the NDVI (Fig. 2.6A), and the inclusion of shape indices (Fig. 2.6C) resulted in less

Table 2.1: Wetland and resulting watershed areas (ha).

Wetland	Wetland area	Watershed area
Humedal de las Salinas del Rasall	26.3	236
Saladar de Cañada Brusca South	3.8	69.5
Saladar de Cañada Brusca North	17.4	360
La Alcanara	199	6,508
Marina del Carmolí	314	16,923
Playa de la Hita	34.4	2,052.8
Saladar de Matalentisco	10.4	907.6
Saladar de la Boquera de Tabala	36.9	5,819.2
Lopoyo	80	2,783
Ajauque	100	7,792
Derramadores	50	1,963
Sombrerico	3	141

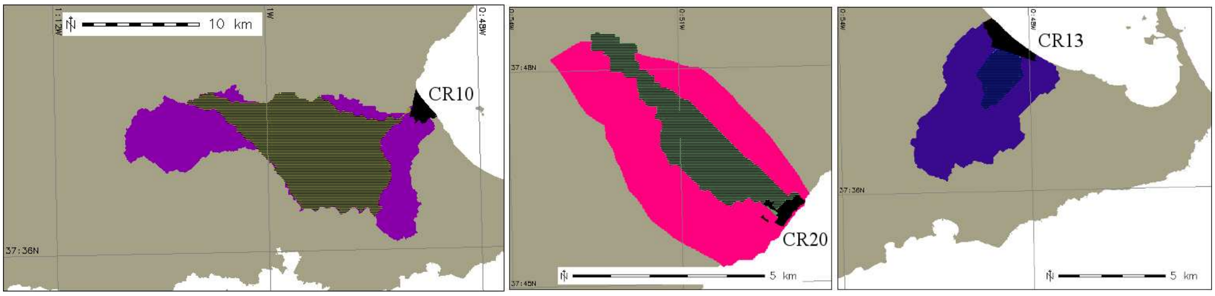

Figure 2.5: Resulting wetland watersheds in the Campo de Cartagena coastal plain. Wetland areas are represented in black. Resulting watershed areas obtained without DEM preprocessing are represented with dashed lines and final watershed areas (using DEM preprocessing) are in different colors. Left: Marina del Carmolí wetland (CR10); center: Lopoyo wetland (CR13); right: Playa de la Hita wetland (CR20).

Table 2.2: Comparison of Overall Accuracy and Kappa coefficients of classifications in 1987 and 2008 after the standard and enhanced classification methodology.

	1987		2008	
	Overall accuracy	**Kappa**	**Overall accuracy**	**Kappa**
Standard methodology	64.87%	0.61	77.19%	0.75
Enhanced methodology	72.51%	0.69	83.01%	0.81

and more compact landscape patches. The 'randomclasiter' method enhanced classification (Fig. 2.6D) and the best results were obtained combining all methodologies (Fig. 2.6E) in terms of overall accuracy and landscape patch identification. Application of the combined enhanced classification method resulted in a 40% reduction in the number of landscape patches in comparison with the standard methodology and maximum patch size increased by three times. Results from the accuracy analysis by land cover class showed that high user's and producer's accuracies were generally obtained in the 1987 and 2008 classifications (table 2.3). As a whole, natural land cover classes (DNW, ONW, DNS and ONS) were accurately assessed in both years, as well as water bodies (WBs). Dry farmed areas (DAC and DHC) were partially overrepresented in the 2008 classification, as well as irrigated crops (IAC and IHC) in 1987. Conversely, greenhouses (GHs) and infrastructure (INF) were slightly underrepresented in the 1987 classification.

2.3.3. Watershed land cover changes

Land-cover maps in 1987 and 2008 were obtained for all wetland watersheds, except for those which were only studied in 2008, *i.e.* Lopoyo (CR13), Derramadores (CR15), Ajauque (CR14) and Sombrerico (CR21) wetlands. Land cover classes of wetland watersheds mapped both years showed an increase in irrigated areas, except for Cañada Brusca South and Rasall wetland watersheds which did not show irrigated areas during the study period (Fig. 2.7). In 1987 dry farmed areas were predominant in the watersheds of Marina del Carmolí, Boquera de Tabala and Alcanara wetlands, while natural areas were the predominant land cover class in the watersheds of Cañada Brusca North and South, Matalentisco and Rasall wetlands. Wetland watersheds mapped only in 2008 showed high percentages of irrigated areas. As an example, resulting land cover maps for the Marina del Carmolí wetland watershed are in figures 2.8 – 2.9.

2.4. Discussion and conclusions

This study proposes enhanced free and open source hydrological modeling and image classification procedures, supporting landscape ecology studies in relation to environmental modeling and monitoring (Newton et al, 2009). Previous studies on wetlands located in the Campo de Cartagena coastal plain did not provide specific watershed areas for the study sites

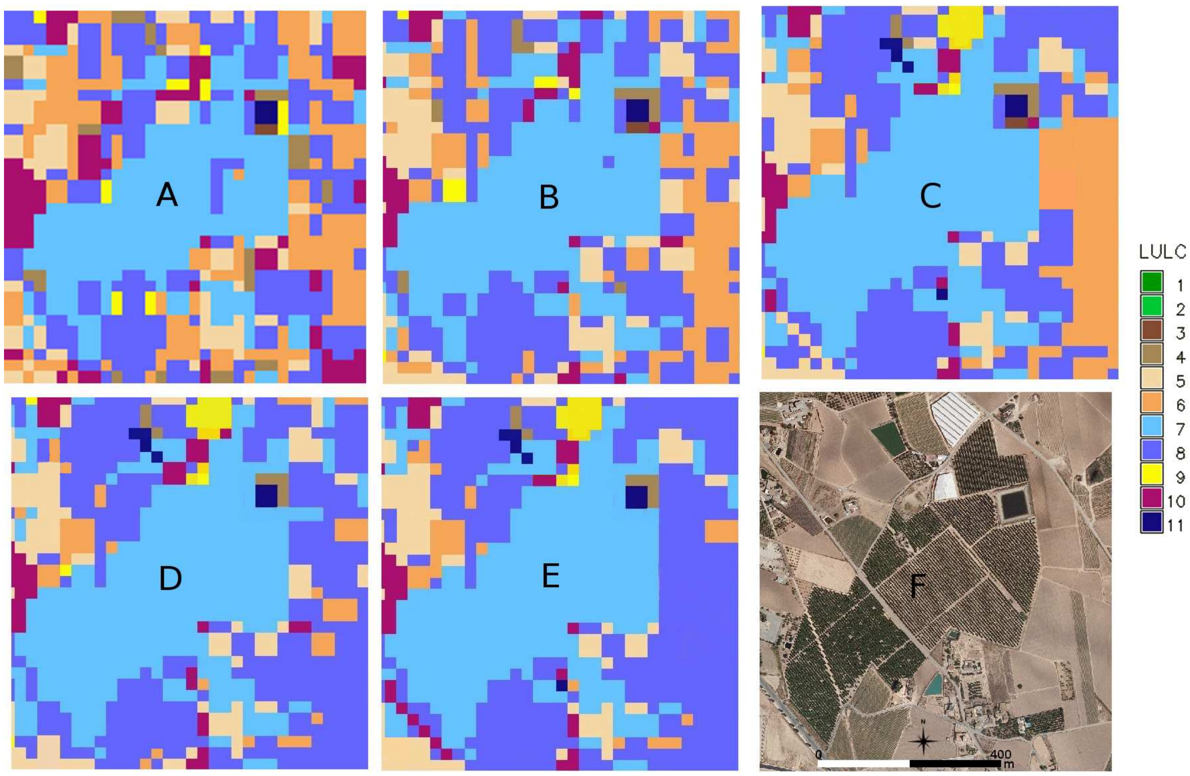

Figure 2.6: Detail of classification maps using different methodologies: (A) standard ML classification method including only NDVI as spectral index, (B) standard ML classification method including all spectral indices, (C) standard ML classification method including spectral indices and contextual information (D) iterative training selection procedure including spectral indices, (E) iterative training selection procedure including spectral indices and contextual information, (F) Aerial picture of the study area. Key of land cover classes: **1:** dense and **2:** open natural woodland; **3:** dense and **4:** open natural shrubland; **5:** dry arboreal and **6:** herbaceous cropland; **7:** irrigated arboreal and **8:** herbaceous cropland; **9:** greenhouses; **10:** unproductive land and infrastructure and **11:** water bodies.

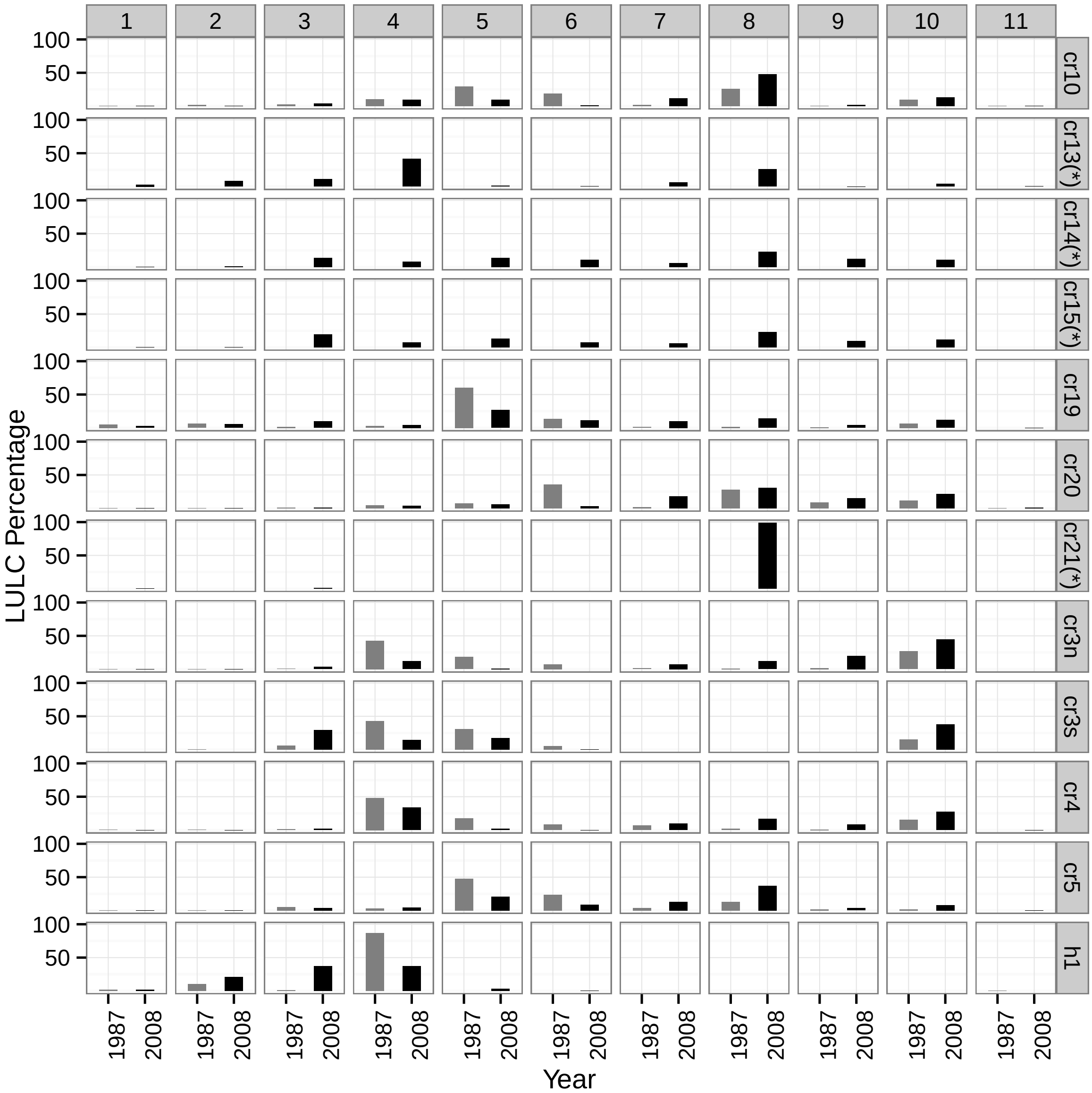

Figure 2.7: Land use / land cover changes in wetland watersheds during the study period. Key of land cover classes: **1:** dense and **2:** open natural woodland; **3:** dense and **4:** open natural shrubland; **5:** dry arboreal and **6:** herbaceous cropland; **7:** irrigated arboreal and **8:** herbaceous cropland; **9:** greenhouses; **10:** unproductive land and infrastructure and **11:** water bodies. Wetland keys: **H1**: Rasall; **CR3N**: Cañada Brusca North; **CR3S**: Cañada Brusca South; **CR10**: Carmoli; **CR5**: Alcanara; **CR14**: Ajauque; **CR13**: Lopoyo; **CR15**: Derramadores; **CR19**: Boquera de Tabala; **CR20**: Playa de la Hita; **CR4**: Matalentisco; **CR21**: Sombrerico. Wetland watersheds denoted with an asterisk show only land cover values for year 2008.

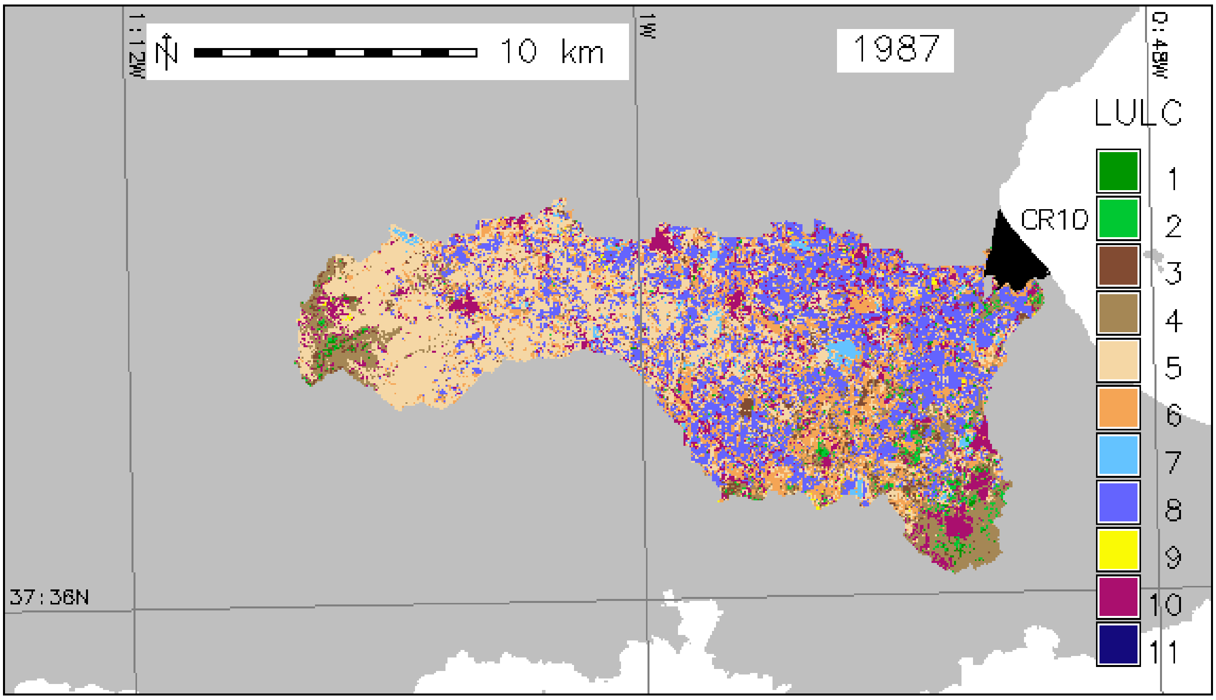

Figure 2.8: Land use / land cover maps in Marina del Carmolí wetland watershed (CR10) in 1987. Key of land cover classes: **1:** dense and **2:** open natural woodland; **3:** dense and **4:** open natural shrubland; **5:** dry arboreal and **6:** herbaceous cropland; **7:** irrigated arboreal and **8:** herbaceous cropland; **9:** greenhouses; **10:** unproductive land and infrastructure and **11:** water bodies.

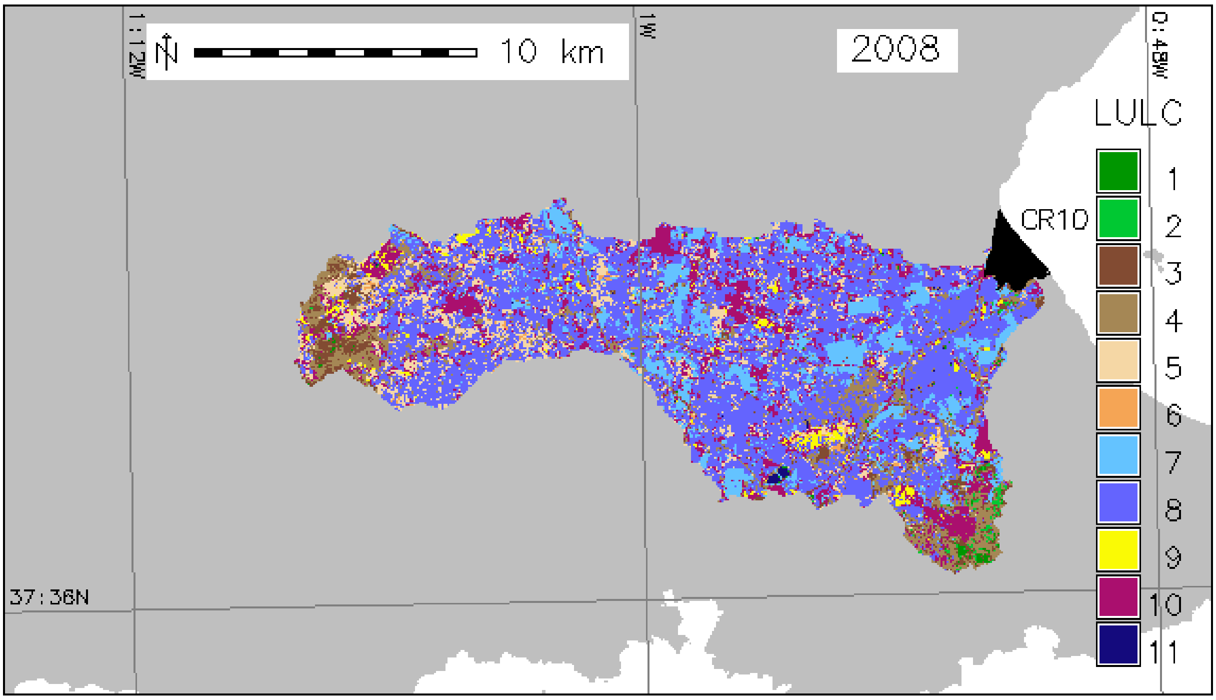

Figure 2.9: Land use / land cover maps in Marina del Carmolí wetland watershed (CR10) in 2008. Key of land cover classes: **1:** dense and **2:** open natural woodland; **3:** dense and **4:** open natural shrubland; **5:** dry arboreal and **6:** herbaceous cropland; **7:** irrigated arboreal and **8:** herbaceous cropland; **9:** greenhouses; **10:** unproductive land and infrastructure and **11:** water bodies.

Table 2.3: Accuracy coefficients (%) of classification maps for each land cover class in 1987 and 2008 using the enhanced methodology. Land cover classes key: dense (DNW) and open (ONW) natural woodland, dense (DNS) and open (ONS) natural shrubland, dry arboreal (DAC) and herbaceous (DHC) cropland, irrigated arboreal (IAC) and herbaceous (IHC) cropland, greenhouses (GHs), infrastructure (INF) and water bodies (WBs).

	1987		**2008**	
	User's accuracy	**Producer's accuracy**	**User's accuracy**	**Producer's accuracy**
DNW	84.62	78.57	80.80	86.79
ONW	71.05	71.05	54.87	86.01
DNS	62.27	75.71	78.24	69.38
ONS	68.99	78.18	70.03	51.32
DAC	90.52	63.25	48.23	81.90
DHC	71.08	70.77	50.79	85.16
IAC	48.33	90.00	94.83	88.75
IHC	62.50	84.75	88.59	69.26
GHs	80.00	57.14	91.59	93.90
INF	95.65	57.89	91.20	95.21
WBs	98.32	98.67	100.00	98.76

but rather focused on the whole Campo de Cartagena catchment area (Carreño et al, 2008; Esteve et al, 2008). However, this methodology was successfully applied and tested in several studies allowing the assessment of specific pressures for each wetland site studied (see chapters 3 and 4).

Following the proposed methodology, landscape patches are much better identified and higher accuracies are obtained in the resulting land use/land cover maps in comparison with the standard supervised classification. However, standard accuracy measures might not sufficiently reflect these enhancements since they are pixel based (Duro et al, 2012). The applied hydrological correction methods successfully improved calculation of watershed areas in wetlands located in plain areas and for some complex wetlands like Cañada Brusca, which receive different surface hydrological influences. As a way to resolve flow direction in flat areas, the DEM stream burning pre-processing together with the creation of sinks within wetland areas served for the proper delineation of wetland watersheds. Even though parallel flow paths were obtained due to the single flow direction algorithm used (D8), globally the flow went towards the ephemeral river network cells. Therefore, the use of multiple flow direction algorithms, like D-Infinity, could enhace flow path maps in flat areas (Tarboton, 1997).

By means of the 'randomclasiter' procedure, the problem of the representativeness of training sites was minimized, resulting in higher map accuracies. This is especially the case in the 1987 land use/land cover classification maps due to the lower resolution of older aerial photographs, which increased uncertainty in training sites selection. This iterative method contributes to the existing ensemble classification methods, like random forest (Breiman, 2001; Gislason et al, 2006), allowing the use of parametric classifiers and its direct application under the GRASS GIS environment. This improved image classification methodology can be also applied to analyze pixel thematic uncertainties, based on the number of times that a pixel was assigned to each different category, in relation to spatial and spectral variables (Brown et al, 2009).

The use of several spectral indices as ancillary layers enhanced the discrimination of land cover classes and the inclusion of patch scale geometric attributes of different land cover classes resulted in more compact landscape patches (Narumalani et al, 1998; Whiteside et al, 2011; Aguirre-Gutiérrez et al, 2012). *Spring* software, used for image segmentation, is proposed as an effective and free alternative to commercial ones (e.g. *eCognition*). In this study, we tried to use a minimum set of spectral indices which targeted general land cover classes like vegetation, urban areas, water bodies and bare soil areas. However, some other spectral indices can be used, such as SAVI (Soil-Adjusted Vegetation Index; Huete 1988) and MSAVI (Qi et al, 1994), especially for semi-arid environments with low vegetation densities, which could enhance classification results.

Overall, irrigated areas increased across wetland watersheds during the study period, except for two of them. Landsat is an appropriate sensor for long term historical studies due to its several satellite missions since 1972 (Wolter et al, 2006). Spatial, temporal and spectral resolution of Landsat sensors were also suitable for land cover mapping at a regional scale (Franklin and Wulder, 2002).

Results promote the use of free and open source software for these kinds of studies, due to their applicability and worldwide free availability (Tufto and Cavallini, 2005; Steiniger and Hay, 2009). This methodology can be easily further developed and tested with new satellite sensors like the Landsat Data Continuity Mission (Irons et al, 2012) and open source image processing algorithms implemented in some libraries in python (Scikit-image Team, 2013) and R (Goslee, 2011). Furthermore, the proposed method is easily automated by means of bash scripts, thus resulting in a free and effective procedure for wetland watershed assessment and monitoring.

3 Wetland and landscape indices for assessing the condition of semiarid Mediterranean saline wetlands under agricultural hydrological pressures

Abstract

During last decades semiarid Mediterranean saline wetlands have undergone several hydrological and biological changes as a consequence of increased water inputs from agricultural areas. Specific indices are needed in order to assess the condition of these unique ecosystems in relation to major hydrological disturbances at watershed level. Through the long-term study of selected plant taxa in a set of representative wetlands in Murcia province (SE Spain), together with the characterization of their watershed agricultural land uses, plant indicators of wetland condition were sought and then combined into a wetland condition index. Percentages of land cover classes of interest were weighted taking into account land cover arrangement and receiving wetland size. Characteristic perennial plant taxa were sampled in 1989 and 2008 and significant taxa frequency changes at each wetland site were determined. Regression analysis was used to relate wetland plant taxa frequency and watershed condition during the study period. *Limonium* spp., *Arthrocnemum glaucum*, *Phragmites australis*, *Tamarix canariensis* and *Atriplex halimus* showed significant relationships with watershed condition. Indicator taxa were thus identified and their frequencies were combined into an integrated index of wetland condition, which showed a robust relationship with watershed hydrological condition.

3.1. Introduction

Wetlands naturally act as sinks of surface and subsurface drainage effluents due to their relative low position in the landscape, thus reflecting upland occurring processes. In last decades, especially in coastal plain areas, there has been a flourishing growth of agricultural irrigated land areas in many semiarid Mediterranean regions (Herrero and Snyder, 1997; Levin et al, 2009; Martínez-Fernández et al, 2005). More specifically, in Murcia province the opening of the Tagus-Segura river water transfer in 1979 promoted the conversion of most dry farmed lands into irrigated land areas. The current expansion of agricultural irrigated lands at watershed scale has led to significant hydrological imbalances that alter wetland ecosystems, thus threatening its conservation (Castañeda and Herrero, 2008a; Esteve et al, 2008; Ortega et al, 2004).

Monitoring actions applied to management and conservation of wetlands require precise indicators in order to obtain accurate information about ecosystem state and functioning and provide an effective early warning system (Fancy et al, 2009). Although much effort has been applied towards protection of wetlands, the preservation of surrounding areas in which they are embedded has been ignored (Houlahan and Findlay, 2004). Assessment of watershed hydrological condition plays therefore a vital role in wetland management (Mack, 2006; Turner et al, 2003; Wigand et al, 1999).

The European Water Framework Directive (European Commission, 2000) seeks the development of indicators of ecological status for freshwater ecosystems, specifically including wetlands (European Commission, 2003). However, most indicators established for aquatic ecosystems are not suitable for semiarid Mediterranean saline wetlands, also called crypto-wetlands (Carreño et al, 2008; Innis et al, 2000; Williams, 1999). These are semi-terrestrial environments between steppes and standing water ecosystems (Vidal-Abarca et al, 2003; Castañeda and Herrero, 2008b).

The development and selection of ecological indicators is a complex process for which different approaches can be used (Carignan and Villard, 2002; Niemeijer and Groot, 2008). Physico-chemical indicators of wetland habitat condition can be very labor intensive and may not necessarily be biologically relevant (Gergel et al, 2002). However, biotic indicators, and specifically plants, may integrate changes in wetland condition over time, may be easy to identify and may provide information on the type of pressures if their ecological tolerances are known (Cronk and Fennessy, 2001; Mitsch and Gosselink, 2007).

Plant species diversity of semiarid saline wetlands is relatively low and it is differentiated according to the plants' individual tolerance of salinity, fluctuating water table levels and anoxic substrate. The establishment of plants as ecological indicators comprises the study of species responses to a known range of a given environmental stressor (Allan, 2004; Niemi and McDonald, 2004). In this regard, results arising from the study of wetland plant species and surrounding land uses at single dates may not be representative of the whole range of species responses and environmental gradients. It has been suggested, for example, that there is a delayed response of the biota to certain landscape environmental variables (Carreño et al, 2008; Harding et al, 1998). Besides, species might still occur in wetlands long after the conditions that promoted their presence have vanished. Therefore, interpretation of site and date specific data needs to be conducted within the larger spatial and temporal perspective (Álvarez Rogel et al, 2006; Dale and Beyeler, 2001).

Although several indices based on plant species and communities have previously been used as a tool for wetland condition assessment in the U.S.A. (Johnston et al, 2009; López and Fennessy, 2002; Miller et al, 2006), such indices are lacking for semiarid Mediterranean saline wetlands and are needed in order to fulfill the European Water Framework. In the context of a proposal for monitoring semiarid Mediterranean saline wetlands, the main aim of this study was to investigate plant taxa as an ecosystem attribute that reflects long-term changes in wetland hydrological conditions.

More specifically, the objectives were (1) to assess major changes in wetland plant taxa composition and in their associated watershed land cover classes in a set of representative semiarid saline wetlands over a 20 years period, (2) to characterize watershed hydrological condition for each wetland in relation to agricultural hydrological pressures, (3) to establish relationships between wetland plant taxa and watershed hydrological condition, and (4) to develop and validate a wetland condition index based on the resulting indicator plant taxa. To accomplish this, historical fieldwork sampling, remote sensing and hydrological modeling techniques were combined.

3.2. Methods

3.2.1. Study wetlands

Eight representative wetlands from which we had plant records in 1989 and 2008 were selected, *i.e.* 6 coastal and 2 inland wetlands (figure 3.1 and table 3.1). Selected sites are included in the regional inventory of wetlands (Vidal-Abarca et al, 2003) and their protection status ranges from regional, national to international level due to their high ecological values (Ramsar Site, Special Protection Area for Birds, Site of Community Importance and Special Protection Area for the Mediterranean), except for Matalentisco and Boquera de Tabala wetlands, which do not benefit from any protection status. Marina del Carmolí and Playa de la Hita wetlands are in a lowland coastal plain, called Campo de Cartagena, associated with the internal shore of the Mar Menor coastal lagoon, which comprises 12,700 ha (Conesa, 1990; Conesa and Jiménez-Cárceles, 2007). Salinas del Rasall is a coastal wetland associated with a salt extraction pond embedded in the Calblanque Natural Park, protected in 1987 by the regional authorities and also included in the Mar Menor RAMSAR protected area. Matalentisco and Cañada Brusca North and South are coastal wetlands located in the southern part of the region on the Mediterranean Sea. Boquera de Tabala and Alcanara are inland wetlands associated with an ephemeral river and with a saline alluvial plain, respectively. Salinas del Rasall and Cañada Brusca South wetlands were selected as reference sites, as their watersheds did not present irrigated agricultural land areas during the study period.

Characteristic plant communities of these wetlands are salt steppes and salt marshes, which occupy areas with high salinity conditions and low and high water availability, respectively. The salt steppe units are composed of the priority habitat 1510 "Mediterranean salt steppes, Limonietalia" and habitat 1430 (Halo-nitrophilous scrubs Pegano-Salsoletea) of the European Habitat Directive (European Commission – Directorate General for Environment, 2007). Main species in salt steppe are *Lygeum spartum, Suaeda vera, Frankenia corymbosa* and *Limonium* spp. The salt marsh units are dominated by habitat 1420 (Mediterranean and thermo-Atlantic halophilous scrubs, Sarcocornetea fruticosi) and habitat 1410 (Mediterranean salt meadows). Main species in saltmarsh are *Sarcocornia fruticosa, Arthrocnemum glaucum, Halimione portulacoides and Halocnemum strobilaceum*. Habitat 92D0 (Southern riparian galleries and thickets) is also represented in sandy areas, composed of *Tamarix canariensis* and *Tamarix boveana*. Finally, the reed beds unit, when present, is dominated by *Phragmites australis* and

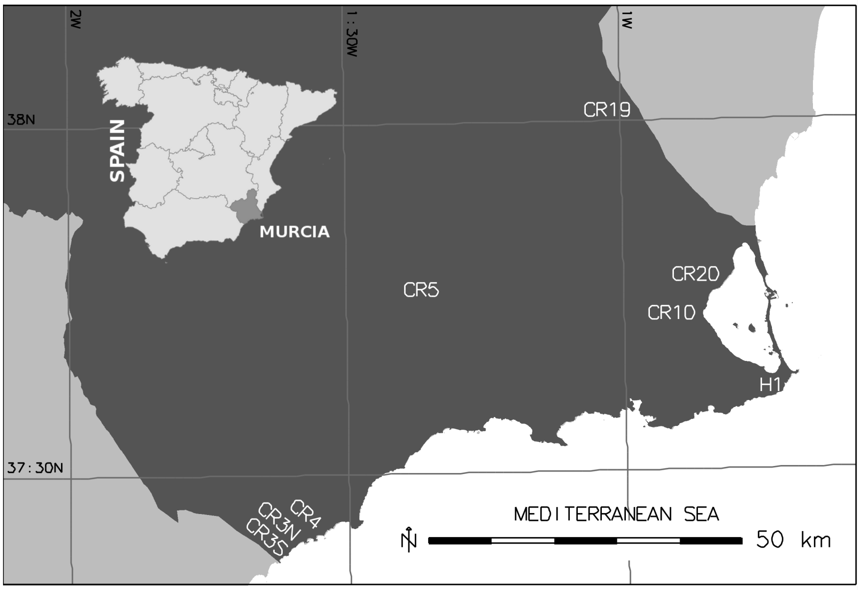

Figure 3.1: Location map of the study wetlands in Murcia province. Wetland keys: Cañada Brusca North (CR3N) and South (CR3S), Alcanara (CR5), Marina del Carmolí (CR10), Playa de la Hita (CR20), Matalentisco (CR4), Boquera de Tabala (CR19) and Salinas del Rasall (H1).

occupies areas with regular water flows and lower salinity (Álvarez Rogel et al, 2006, 2007c; MARM, 2009). All habitats, excluding reed beds, are recognized as being of Community Interest by the Habitat Directive and habitat 1510 (Mediterranean salt steppes) is designated of Priority Interest.

3.2.2. Watershed delineation

Specific watershed boundaries for each wetland were delineated from a 10 m raster digital elevation model (DEM) using single flow direction method (D8 algorithm; O'Callaghan and Mark, 1984). Prior to watershed delineation, the DEM was modified in the Campo de Cartagena coastal plain area by lowering the elevation values coinciding with existent stream network to force flow-direction models to match existing stream lines (King et al, 2005; Strayer et al, 2003). Since watersheds are usually delineated by the area upstream from a given outlet point, DEM elevation values within larger wetlands in the Campo de Cartagena area were modified by creating an artificial sink in order to force all flow-accumulation cells draining into the wetland to converge into a single cell. All GIS analyses were performed with GRASS GIS 6.4 (GRASS Development Team, 2008). For more details, see chapter 2. Wetland watersheds ranged from 70 to 17,000 hectares (table 3.1).

Table 3.1: Characterization of wetlands and their associated watersheds.

Name	Wetland size (ha)	Watershed size (ha)	Location	Reference site
Salinas del Rasall	26.3	236	coastal	yes
Cañada Brusca South	3.8	69.5	coastal	yes
Cañada Brusca North	17.4	360	coastal	no
Marina del Carmolí	314	16,923	coastal	no
Playa de la Hita	34.4	2,052.8	coastal	no
Matalentisco	10.4	907.6	coastal	no
Boquera de Tabala	36.9	5,819.2	inland	no
Alcanara	199	6,508	inland	no

3.2.3. Land use/land cover mapping

Historical and medium spatial resolution land cover data were needed to assess specific pressures on the study wetlands over time. For each wetland watershed, land cover maps in years 1987 and 2008 were obtained by means of remote sensing techniques. Supervised classification of eleven land cover classes was performed using the widely used maximum likelihood algorithm (Michelson et al, 2000; Richards and Jia, 2006). Landsat images sensors TM and ETM+ were used for years 1987 and 2008, respectively, and image pixel size was set to 25 m. Each classification was carried out with two images (summer and winter), including single bands and diverse spectral and patch shape indices as ancillary layers: Normalized Difference Vegetation Index (NDVI; Bannari et al, 1995), Modified Normalized Water Index (MNDWI; Hui et al, 2008), Normalized Difference Built-up Index (NDBI; Zha et al, 2003) and Normalized Difference Bareness Index (NDBaI; Chen et al, 2006), as well as patch fractal dimension (FD) and patch shape indices (SI) (Chust and Ducrot, 2004). Land use patches were previously extracted by means of automatic image segmentation using the region growing algorithm with SPRING software (Cāmara et al, 1996). Training sites were obtained using aerial photos from 1987 and 2008 of 1 m and 0.45 m resolution, respectively. The classification method was enhanced by an iterative training sites random selection method (Carreño et al, 2011; González, 2011). The resulting maps were verified by visual validation on aerial photos using a stratified random sampling. For more details, see chapter 2.

Seven land cover classes were studied: dense and open natural woodlands, dense and open natural shrublands, rainfed tree and herbaceous croplands, irrigated tree and herbaceous croplands, greenhouses, urban areas and water bodies. Urban areas and water bodies were not further considered in subsequent analyses, since they represented low percentages in the studied watersheds. Rainfed cropland areas were also disregarded since they were historically present and their water outputs are negligible in comparison to those from irrigated cropland areas (Conesa, 1990; Velasco et al, 2006). The rest of land cover classes were pooled into two categories: natural areas (NAT), including dense/open natural woodlands and dense/open natural shrublands, and irrigated agricultural land areas (ILA), including irrigated tree and herbaceous croplands and greenhouses. Land cover maps showed an overall accuracy percentage of 73% and 83% for 1987 and 2008 classification, respectively. There was a significant and a marginally significant inverse Pearson correlation between natural and irrigated land areas in wetland watersheds, being higher in 2008 ($r = -0.83; P = 0.01$) than in 1989 ($r = -0.69; P = 0.06$).

3.2.4. Watershed hydrological condition index

In order to assess and compare the hydrological pressures that irrigation at watershed scale exerts on different wetlands, raw percentages of irrigated and natural land cover classes in the watersheds were weighted by landscape factors. First, since the irrigation flows from near irrigated areas were supposed to exert more influence on wetland hydrology (Castañeda and Herrero, 2008b), we used inverse-distance weights (IDW) to characterize land-cover arrangement within watersheds (King et al, 2005; Van Sickle and Johnson, 2008). Secondly, since wetland sizes differed over two orders of magnitude, ranging from 3.8 to 314 hectares, we considered that the effect of drainage inputs on larger wetlands should be considerably lower than in smaller ones. Thus, IDW percentages of natural and and irrigated land areas in the watershed were divided by the receiving wetland area and then square rooted, according to a wetland area relative percentage index (WARP; equation 3.1).

$$\mathrm{WARP}_{LC} = \sqrt{\frac{\text{LC (IDW) in watershed (\%)}}{\text{Wetland area (ha)}}} \tag{3.1}$$

Where LC refers to a specific land cover of interest (natural and irrigated land areas), which were first inverse-distance weighted (IDW).

3.2.5. Vegetation sampling

To characterize plant taxa composition at each wetland site, frequency of nine dominant characteristic perennial plant taxa was recorded in years 1989 and 2008 by means of field sampling. Sampled taxa were: *Arthrocnemum glaucum*, *Atriplex glauca, Atriplex halimus, Limonium* spp. (including *Limonium cossonianum, Limonium caesium* and *Limonium insigne*), *Tamarix canariensis, Phragmites australis, Suaeda vera, Sarcocornia fruticosa* and *Halimione portulacoides*. The number of sampling units in each wetland ranged from 12 to 116, depending on wetland size and heterogeneity of plant communities. Sampling units consisted of 25 m^2 circular areas regularly distributed. Presence/absence of selected taxa was recorded at each sampling unit and overall taxa frequencies, ranging from zero to one, were then calculated for each wetland. Wetland areas remained constant throughout the study period except for Boquera de Tabala wetland, which was slightly reduced in size during the study period due to road works.

3.2.6. Selection of indicator plant taxa

Taxa showing significant frequency changes at each wetland site over the study period were first identified using a pairwise comparison binomial test (Crawley, 2007) in order to select potential indicator taxa, whose frequencies might be linked to watershed hydrological condition.

Subsequently, final indicator taxa were selected by means of linear regression analyses between their frequency and watershed hydrological condition in years 1989, 2008, and the observed changes during the study period. Taxa frequencies were arcsine square root transformed to accomplish normality prior to regression analysis (Logan, 2010). Careful inspection of residual plots was performed in order to check for normality and independence of residuals. Data were analysed with the statistical package R (R Core Team, 2013).

3.2.7. Wetland condition index

Frequencies of resulting indicator taxa were combined as an index to assess wetland condition (WCI) in relation to watershed hydrological pressures. For this purpose, the sum of taxa frequencies that were positively related to hydrological alterations (considered as negative indicator taxa) was substracted to the sum of taxa frequencies that were positively related to naturalness (considered as positive indicator taxa). Total frequency of positive indicator taxa was square rooted in order to increase their weight at low occurring frequencies, while diminishing frequency values higher than one. The fact that both positive indicator taxa were occurring at high frequencies, could probably be more related to wetland natural heterogeneity than to its condition status. On the contrary, the sum of negative indicator taxa was squared in order to underpin the presence of more than one negative indicator taxon (sum of values higher than one), while minimizing the effect of total frequencies below one (equation 3.2).

$$WCI = \sqrt{\text{Total freq. positive indicator taxa}} - (\text{Total freq. negative indicator taxa})^2 \tag{3.2}$$

In order to test the applicability of the proposed index in a wider set of wetlands, four additional wetlands (Sombrerico, Lo Poyo, Ajauque and Derramadores) were selected in Murcia province, for which the same taxa and watershed information was recorded and calculated for year 2008. Finally, the resulting index was tested against watershed hydrological condition by means of regression analysis.

3.3. Results

3.3.1. Watershed hydrological condition index

Overall, irrigated agricultural land areas increased across wetland watersheds during the study period. While the cover of irrigated land areas (ILA-IDW) represented less than 40% in 1987, it reached values over 60% in 2008 across wetland watersheds (figure 3.2). While Cañada Brusca North watershed showed the highest increase in irrigated land areas during the study period, they were absent in Cañada Brusca South and Salinas del Rasall watersheds during the study period (our reference wetlands). After wetland area relative percentages (WARP) were calculated, larger wetlands showed relatively lower irrigated agricultural land areas values than smaller ones.

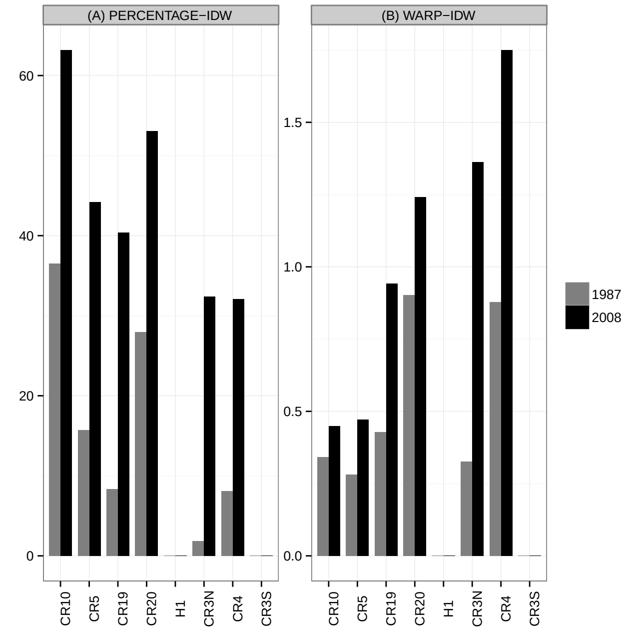

Figure 3.2: Irrigated agricultural land areas (ILA) in wetland watersheds in 1987 and 2008. Figures 3.2A and B show the percentage of irrigated land areas (inverse distance weighted: IDW) and the wetland area relative percentages (WARP-IDW), respectively. Wetland keys: Cañada Brusca North (CR3N) and South (CR3S), Alcanara (CR5), Marina del Carmolí (CR10), Playa de la Hita (CR20), Matalentisco (CR4), Boquera de Tabala (CR19) and Salinas del Rasall (H1). Wetlands are arranged from larger to smaller sizes from left to right.

3.3.2. Vegetation changes and selection of indicator taxa

The binomial paired test showed a total of 24 significant changes in plant taxa frequency across wetlands during the study period (figure 3.3). Taxa occurring at a specific wetland with frequencies lower than 0.05 in both years were disregarded from further analyses.

Cañada Brusca North underwent the highest number of significant changes in taxa frequency, while Cañada Brusca South and Salinas del Rasall wetlands (our reference sites) showed very few significant changes in taxa frequency during the study period. Across wetlands, a decreasing trend in frequencies of *Limonium* spp., *Arthrocnemum glaucum* and *Atriplex glauca* was observed (figure 3.3). On the contrary, frequencies of *Phragmites australis*, *Tamarix canariensis* and *Atriplex halimus* generally increased in the study wetlands. Frequencies of *Suaeda vera* and *Sarcocornia fruticosa* did not show a consistent pattern of changes across sites. *Halimione portulacoides* showed no significant frequency changes during the study period and hence it was disregarded as a potential indicator taxon.

Final indicator taxa of watershed agricultural pressures were identified by means of regression analysis. Five out of the eight potential indicator taxa showed significant relationships with weighted land cover percentages (figure 3.4). While *Arthrocnemum glaucum* frequency was positively related to natural areas in watersheds in 1989 and 2008 (figure 3.4A; 3.4B), observed frequency changes in *Atriplex halimus* and *Tamarix canariensis* were inversely correlated with changes in natural areas during the study period (figure 3.4D; 3.4F). On the other hand, *Limonium* spp. showed a negative relationship with irrigated land areas in 2008 (figure 3.4C), while *Pragmites australis* frequency changes were positively related to changes in irrigated land areas during the study period (figure 3.4E).

3.3.3. Wetland condition index

The developed wetland condition index was finally proposed as follows (equation 3.3):

$$WCI = \sqrt{Lspp + Agu} - (Pau + Aha + Tca)^2 \tag{3.3}$$

Where $Lspp$ stands for frequency of *Limonium* spp., Agu for *Arthrocnemum glaucum*, Pau for *Phragmites australis*, Aha for *Atriplex halimus* and Tca for *Tamarix canariensis*. The wetland condition index ranged from -9 up to 1.4 (figure 3.5), standing negative values for highly altered wetlands. The proposed WCI showed the highest values at the reference wetlands in 2008 (Salinas del Rasall and Cañada Brusca South), while Cañada Brusca North showed the highest decrease during the study period. Overall, except for for Salinas del Rasall wetland, higher values of WCI were observed in 1989 than in 2008.

In 2008, wetland condition index values, including the additional sites, were negatively related to irrigated land areas in watersheds (WARP-IDW) and to observed changes during the study period (figure 3.6A; 3.6C). The resulting linear model for 2008 was robust, despite of including extreme values of high watershed hydrological pressures and low wetland condition index, as is the case for Sombrerico wetland (figure 3.6B).

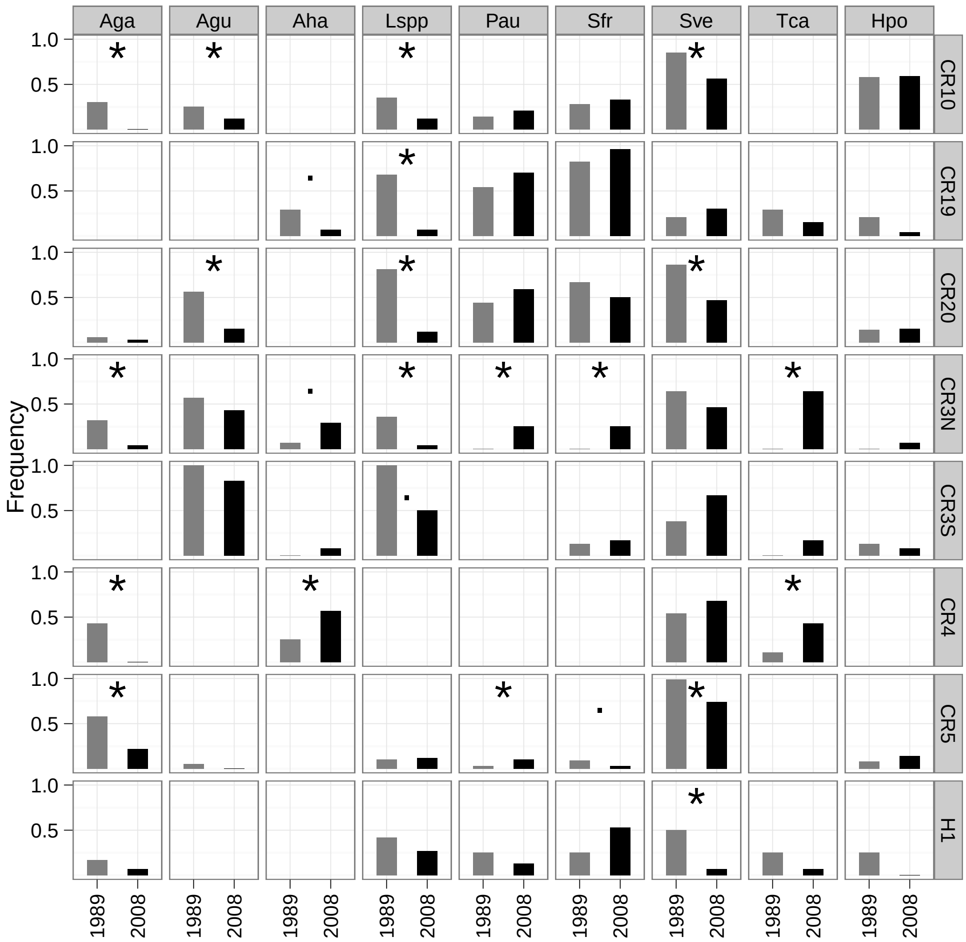

Figure 3.3: Frequency of sampled taxa in 1989 and 2008 at each wetland site. Changes at a significance level of $P < 0.1$ after the pairwise comparison binomial test are denoted with a dot, and changes at significance level of $P < 0.05$ with an asterisk. Taxa keys: *Atriplex glauca* (Aga), *Arthrocnemum glaucum* (Agu), *Atriplex halimus* (Aha), *Limonium* spp. (Lspp), *Phragmites australis* (Pau), *Sarcocornia fruticosa* (Sfr), *Suaeda vera* (Sve), *Tamarix canariensis* (Tca) and *Halimione portulacoides* (Hpo). Wetland keys: Cañada Brusca North (CR3N) and South (CR3S), Alcanara (CR5), Marina del Carmolí (CR10), Playa de la Hita (CR20), Matalentisco (CR4), Boquera de Tabala (CR19) and Salinas del Rasall (H1).

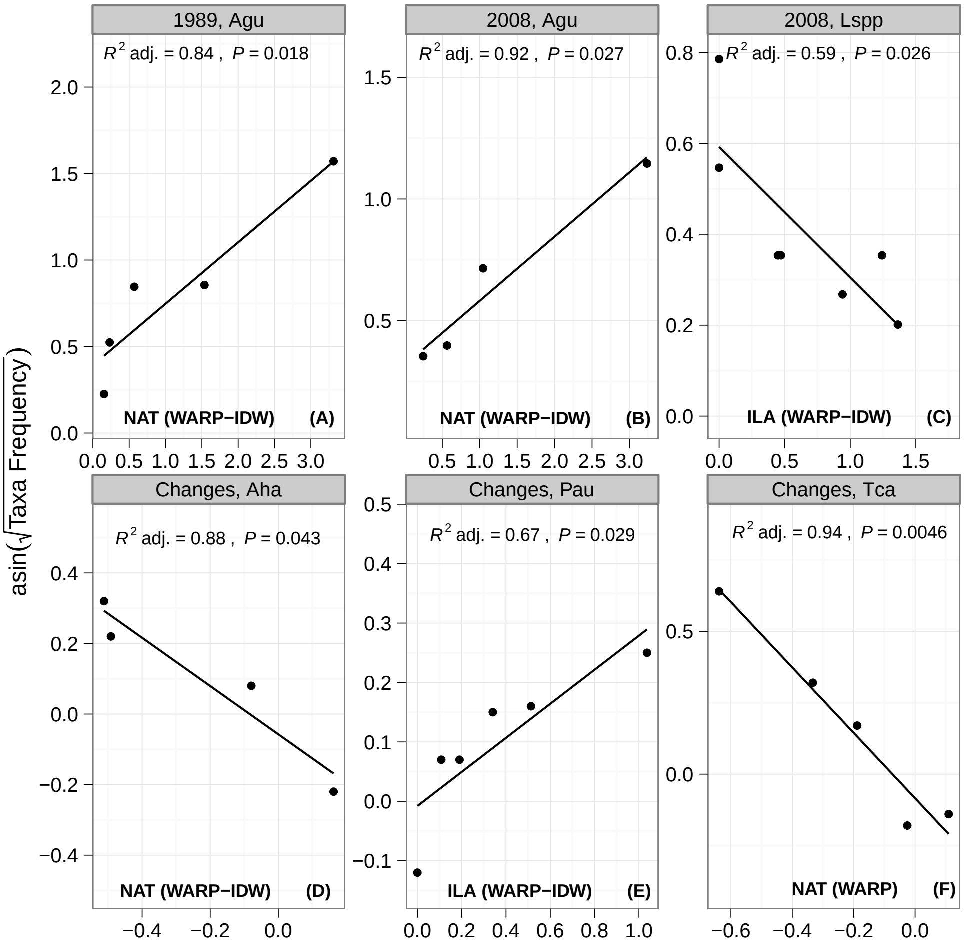

Figure 3.4: Significant linear relationships between taxa frequency and weighted land cover percentages (natural and irrigated land areas) in 1989, 2008 and observed changes: (A) and (B) *Arthrocnemum glaucum* (Agu), (C) *Limonium* spp. (Lspp), (D) *Atriplex halimus* (Aha), (E) *Phragmites australis* (Pau), and (F) *Tamarix canariensis* (Tca).

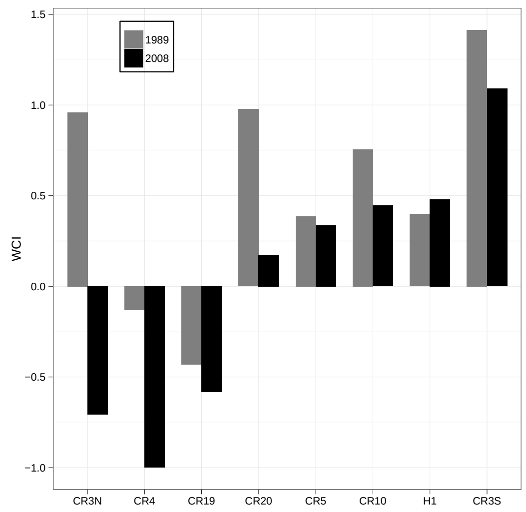

Figure 3.5: Wetland condition index (WCI) values for each wetland in 1989 and 2008. Wetlands are arranged from left to right according to a decreasing gradient of irrigated agricultural land areas changes in their watersheds during the study period. Wetland keys: Cañada Brusca North (CR3N) and South (CR3S), Alcanara (CR5), Marina del Carmolí (CR10), Playa de la Hita (CR20), Matalentisco (CR4), Boquera de Tabala (CR19) and Salinas del Rasall (H1).

3.4. Discussion and conclusions

The proposed wetland condition index (WCI) allows the assessment of Mediterranean semiarid saline wetlands located in agricultural catchments. Inverse distance weighting (IDW) and wetland area relative percentage index (WARP) were useful weighting factors for the assessment of watershed condition in relation to agricultural hydrological pressures on wetlands. Accurate and medium resolution assessment of land cover types and the delimitation of specific watershed areas for each wetland was highly important, as it has been pointed out in previous studies (McHugh et al, 2007; Roth et al, 1996). This was especially important for some complex wetlands like Cañada Brusca, whose fragments (North and South) receive clearly different surface hydrological influences. Moreover, our study revealed that these wetlands showed the most contrasting results in terms of taxa changes during the study period (Figure 3.3).

Salinas del Rasall was the only wetland in which WCI increased during the study period, which might be related to the fact that it was legally protected for four years before our study started (Figure 3.5). On the contrary, WCI decreased in Cañada Brusca South during the study period, probably due to marginal influences coming from the nearby located watershed of Cañada Brusca North, which suffered the highest ILA increase, followed by Matalentisco wetland (Figure 3.2). The fact that Matalentisco and Boquera de Tabala wetlands showed negative WCI values in 1989, might be related to their lack of protection status during the study period. The proposed wetland condition index showed a robust relationship with ILA

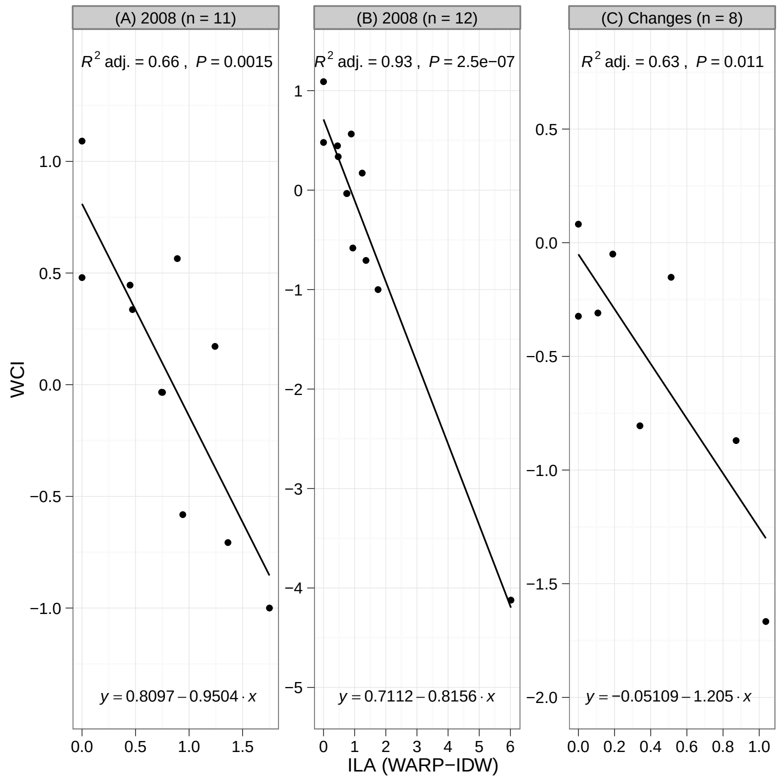

Figure 3.6: Linear regression between the wetland condition index (WCI) and irrigated land areas (ILA WARP-IDW) values. Figures 3.6A and 3.6B show the resulting models in 2008 excluding ($n = 11$) and including ($n = 12$) Sombrerico wetland, respectively. Figure 3.6C shows the resulting model based on observed changes during the study period ($n = 8$).

(WARP-IDW) values in 2008 on an extended set of wetlands (Figure 3.6B). Observed changes in WCI and in ILA (WARP-IDW) in their watersheds during the study period were also significantly related (Figure 3.6C).

Historical information reflecting ecosystem responses to land cover changes enhanced the indicator selection process. Undisturbed or reference sites (present or past) were needed in order to establish reference conditions, despite wetlands natural variability. Although our study was constrained by the number of wetlands sampled in 1989, it was based on a long-term period of time and on a set of heterogeneous wetlands, embedded in a representative range of hydrologically altered watersheds. In 1989, irrigated land areas were not the dominant land cover types (Figure 3.2), therefore we expected lower influence of watershed land cover on wetland plant taxa at that time. On the contrary, the land use intensification gradient across watersheds was large enough to show relationships with some wetland plants in 2008. Moreover, the analysis of land cover and taxa frequency changes revealed which taxa benefited from the expansion of irrigated land areas during the study period.

The expansion of irrigated land areas at watershed scale seems to have altered wetlands natural hydrological regimes, probably in terms of changes in salinity, flooding frequency and nutrient inputs from agricultural runoff (Robledano-Aymerich et al, 2010). Salinity conditions and seasonal hydrological fluctuations might have decreased across the study wetlands, thus negatively affecting taxa which are adapted to the natural conditions of these wetland types and promoting some taxa adapted to less saline conditions and/or longer flooding periods.

As it has been pointed out by previous studies, *Arthrocnemum glaucum* is known to require saline and humid conditions, being adapted to seasonal fluctuations of water table levels (Álvarez Rogel et al, 2006; Caballero, 1999; Pujol Fructuoso, 2002). It was the only taxon showing significant relationship with watershed land cover in 1989 (Figure 3.4A), thus probably serving as an early warning indicator of the presence of irrigated land areas in the watershed. *Limonium* spp. belongs to the drier part of the humidity gradient, avoids flooded soils and withstands high salinities (Caballero, 1999; Álvarez Rogel et al, 2000). *Phragmites australis* is considered an invasive species (Álvarez Rogel et al, 2007c; Tulbure et al, 2007) usually forming monospecific stands, reducing species diversity, and therefore indicating wetland degradation. Its frequency increased in most wetlands along with higher percentages of irrigated land areas in the watershed (Figure 3.4E), probably as a consequence of lower salinity and longer flooding periods (Burdick and Konisky, 2003). *Atriplex halimus* occurrence is associated with lower salinity conditions than salt steppe and salt marsh communities in the drier part of the humidity gradient and might be found in association with salt cedar patches (MARM, 2009).

In conclusion, by meand of historical fieldwork sampling, remote sensing and hydrological modeling techniques, a wetland condition index (WCI) was developed, based on plant taxa frequency, that allows the assessment of Mediterranean semiarid saline wetlands located in agricultural catchments, serving as a management tool to preserve their values and associated ecosystem services. The results from this study reinforce the importance of quantifying the influence of landscape level processes for wetland management and conservation, in accordance with previous studies (Rooney et al, 2012). International management efforts for protecting freshwater ecosystems in Mediterranean areas, like the European Water Framework Directive (European Commission, 2000), the European Habitats Directive (Council of Europe, 1992) and the RAMSAR convention (Ramsar Convention Secretariat, 2004; Finlayson, 2005), should specifically take into account catchment-scale hydrological influences of agricultural land uses on wetlands.

4 Remote sensing of plant communities as a tool for assessing the condition of semiarid Mediterranean saline wetlands in agricultural catchments

Abstract

Semiarid Mediterranean saline wetlands are unique ecosystems sheltering high biodiversity. In the last decades, the expansion of irrigated lands has led to hydrological imbalances in Mediterranean catchments, causing wetland degradation. Vegetation composition assessment is considered an important tool for evaluating wetland ecological condition and can be mapped using remote sensing. This study aims to develop a condition index based on plant community composition suitable for semiarid Mediterranean saline wetlands, as well as to test the applicability of airborne multispectral remote sensors for discriminating plant communities. Characteristic plant communities of 12 wetlands were identified by means of ordination and classification analysis of plant taxa cover percentages obtained through fieldwork sampling. An index for assessing wetland ecological condition was developed based on the relationship between wetland plant community composition and watershed hydrological condition. Selected wetland plant communities were then mapped by means of remote sensing techniques using random forest algorithm for supervised classification of airborne images. Following this methodology, remote sensing served as a tool for wetland condition assessment at a regional scale.

4.1. Introduction

Semiarid Mediterranean saline wetlands are semi-terrestrial ecosystems, which yearly undergo dry periods of several months, and shelter a rich, endemic and sensitive biota (Brinson and Malvárez, 2002; Vidal-Abarca et al, 2003). They are sinks of irrigation flows, which makes catchment scale management of vital importance for their conservation (Dudgeon et al, 2006). In the last decades, the expansion of irrigated areas in semiarid Mediterranean catchments has led to altered inputs of water and nutrients to lowland wetlands (Hollis, 1990; Carreño et al, 2008; Martín-Queller et al, 2010), which particularly affect soil salinity, water table level and regime and soil moisture conditions (Álvarez Rogel et al, 2007c). Although much effort has been applied towards protection of wetlands, the preservation of their watershed areas has been largely ignored (Houlahan and Findlay, 2004). Moreover, the lack of systematic monitoring and management procedures for this type of saline aquatic ecosystems is ultimately leading to extensive wetland degradation, and therefore practical tools for assessing ecosystem state and functioning are required in order to orient decision making (Williams, 2002).

Vegetation composition assessment is considered an important aspect for evaluating wetland ecological condition (López and Fennessy, 2002; Miller et al, 2006; García et al, 2009; Caçador et al, 2013). Anthropic or climatic factors that affect wetland plant communities also affect wetland birds and invertebrates communities (Hughes, 2004; Pardo et al, 2008; Robledano-Aymerich et al, 2010). The European Habitats Directive considers salt marsh and salt steppe habitats as important and endangered ones and promotes their preservation (Council of Europe, 1992). Moreover, the European Water Framework Directive also impels to monitor the ecological status of transitional waters ecosystems including wetlands (European Commission, 2003; Ferreira et al, 2007). Plant species and communities can be used as a proxy to assess wetland hydrological perturbations if their ecological tolerances to environmental factors such as salinity and water table level are known (Cronk and Fennessy, 2001). Previous studies in similar wetlands have focused on individual taxa rather than on plant communities (Álvarez Rogel et al, 2007c). However, plant communities contain more information than single species and are easier to map by means of remote sensors (O'Connell, 2003; Johnston et al, 2009).

Given that wetlands are often located in remote areas and are difficult to survey, fieldwork sampling methods are labour intensive, especially for large wetland sites and for regional scale assessment. Therefore, most of these ecosystems receive no systematic monitoring (Buchanan et al, 2009). In this regard, remote sensing offers a rapid and cost-effective approach for generating ecologically relevant wetland vegetation maps (Jones et al, 2009; Lee and Yeh, 2009; Poulin et al, 2010). Remote sensing techniques are able to cover large areas, are prone to rapid technical improvements, and can help discriminate different wetland plant communities in order to systematically assess wetland conservation status (Belluco et al, 2006; Wang et al, 2007). Due to pixel size, specific plant species are usually more difficult to discriminate, unless they form dense monospecific stands (*e. g.* reed beds). However, plant communities are composed by similar relative proportions of specific plant species across wetland sites and therefore they tend to show characteristic spectral responses at large patches. Therefore, the establishment of plant communities that can be related to wetland hydrological pressures combined with remote sensing techniques can serve as a tool for wetland management and monitoring at a regional scale (Ozesmi and Bauer, 2002; Xie et al, 2008).

Different methods have been developed for wetland vegetation delimitation from remote sensing imagery (Horning et al, 2010; Friess et al, 2012; Szantoi et al, 2013). Wetlands are very heterogeneous ecosystems containing small vegetation patches with similar spectral responses that might generate spectral confusion (Ozesmi and Bauer, 2002; Baker et al, 2006a). Enhanced classification methods like random forest algorithm or artificial neural networks have shown better performance for these Mediterranean ecosystems (Breiman, 2001; Černá and Chytrý, 2005; Sluiter, 2005). Medium-resolution satellite sensors such as Landsat TM are not suitable for detecting small wetlands or plant community patches (Xie et al, 2008; Adam et al, 2010). Therefore, airborne multispectral sensors are a good source of remotely sensed data for wetland vegetation mapping since they combine high spatial and spectral resolution with acquisition timing (Wang et al, 2007; Klemas, 2011).

The scarcity of available historical data on wetland vegetation due to inaccessibility and high economic cost calls for the use of remote sensing for present and future studies in order to provide a set of multi-temporal images to monitor wetland ecosystems (Klemas, 2001; Carreño et al, 2008; MacKay et al, 2009). Scientific studies bridging the gap between ecology and conservation biology are utmost important in order to influence conservation of aquatic ecosystems (Strayer and Dudgeon, 2010). This study proposes a procedure for a comprehensive study of wetlands and their drainage basins coupling fieldwork and advanced remote sensing techniques in order to evaluate wetland condition based on plant communities. The main objectives of this study were: (1) to characterize plant communities in several representative wetlands under a range of watershed hydrological conditions; (2) to explore the relationships between wetland plant community composition and watershed hydrological pressures; (3) to propose a wetland condition index based on plant community composition; and (4) to test the potential of remote sensing as a tool for assessing wetland condition.

4.2. Methods

4.2.1. Study wetlands

Twelve representative wetlands were selected, *i.e.* 8 coastal and 4 inland wetlands (Figures 4.1 and 4.2; Table 4.1). Selected sites are included in the regional inventory of wetlands (Vidal-Abarca et al, 2003) and their protection status ranges from regional, national to international rules due to their high ecological values (Ramsar Site, Special Protection Area for Birds, Site of Community Importance and Special Protection Area for the Mediterranean), except for Matalentisco and Boquera de Tabala wetlands, which do not benefit from any protection status. Marina del Carmolí, Lo Poyo and Playa de la Hita wetlands are in a lowland coastal plain, called Campo de Cartagena, associated with the internal shore of the Mar Menor coastal lagoon, which comprises 12,700 ha (Conesa, 1990; Conesa and Jiménez-Cárceles, 2007). Salinas del Rasall is a coastal wetland associated with a salt extraction pond embedded in the Calblanque Natural Park, and also included in the Mar Menor RAMSAR protected area. Matalentisco, Sombrerico and Cañada Brusca North and South are coastal wetlands located in the southern part of the region on the Mediterranean Sea. Boquera de Tabala, Ajauque, Derramadores and Alcanara are inland wetlands associated with an ephemeral river and with a saline alluvial plain, respectively.

Main plant communities of these wetlands have been described in section 3.2.1 in chapter 3.

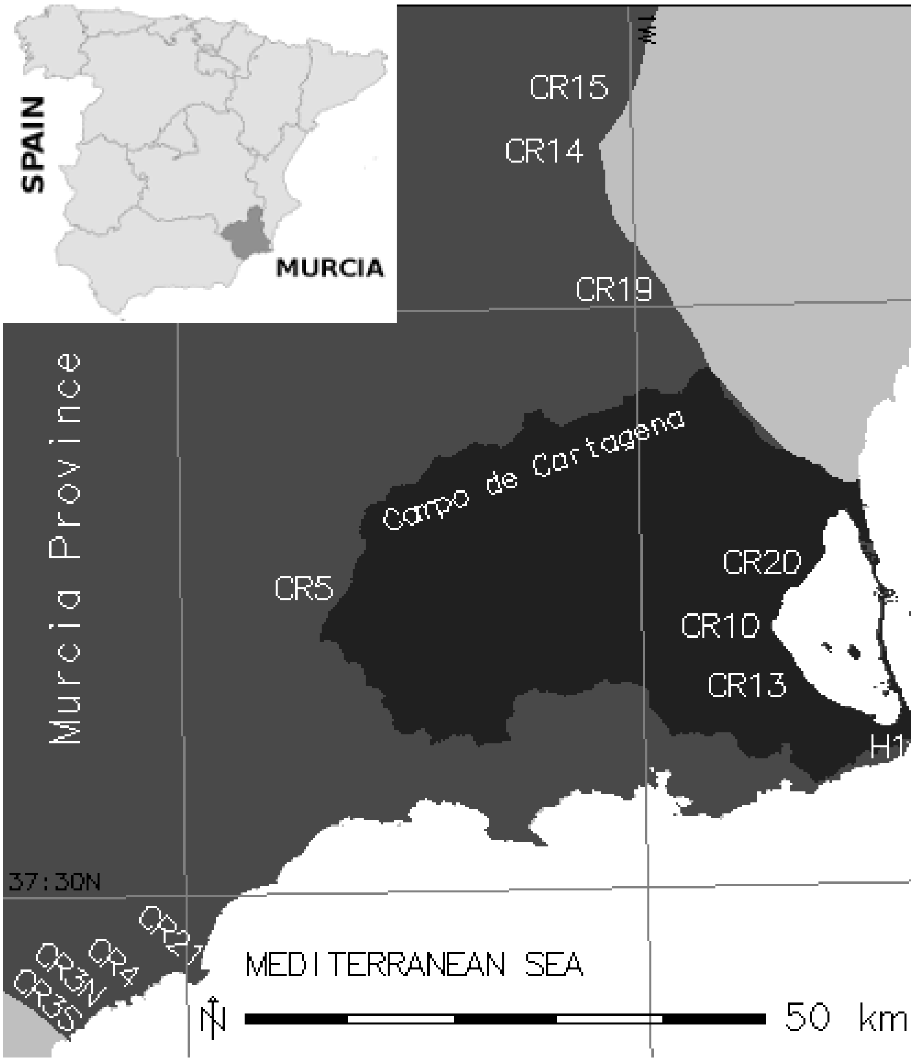

Figure 4.1: Location of Murcia province in Spain and approximate location of the study wetlands. Wetland keys: **H1**: Rasall; **CR3S**: Cañada Brusca South; **CR10**: Carmoli; **CR5**: Alcanara; **CR14**: Ajauque; **CR13**: Lopoyo; **CR15**: Derramadores; **CR19**: Boquera de Tabala; **CR20**: Playa de la Hita; **CR3N**: Cañada Brusca North; **CR4**: Matalentisco; **CR21**: Sombrerico.

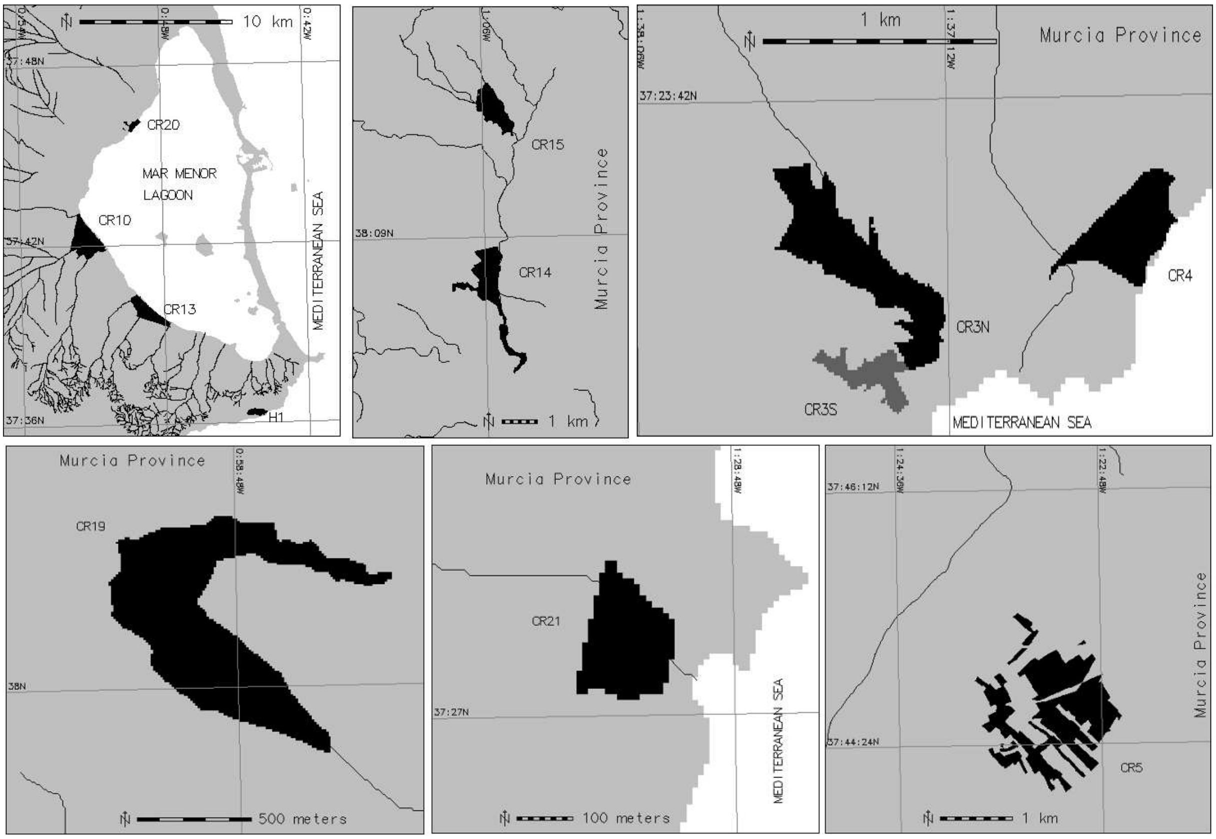

Figure 4.2: Wetlands area maps and relation to the river network. Wetland keys: **H1**: Rasall; **CR3S**: Cañada Brusca South; **CR10**: Carmoli; **CR5**: Alcanara; **CR14**: Ajauque; **CR13**: Lopoyo; **CR15**: Derramadores; **CR19**: Boquera de Tabala; **CR20**: Playa de la Hita; **CR3N**: Cañada Brusca North; **CR4**: Matalentisco; **CR21**: Sombrerico.

4.2.2. Wetland basin characteristics

Specific watershed boundaries for each wetland were delineated from a raster DEM with a pixel size of 10 m using single flow direction method (D8 algorithm). Prior to watershed delineation, the digital elevation model (DEM) was modified in the Campo de Cartagena coastal plain area to enhance the delineation of wetland watersheds. To accomplish this, elevation values coinciding with existent stream network were lowered to force flow-direction models to match existing stream lines (Strayer et al, 2003; King et al, 2005). Since watersheds are usually delineated by the area upstream from a given outlet point, DEM elevation values within larger wetlands in the Campo de Cartagena area were also modified by creating an artificial sink in order to force all flow-accumulation cells draining into the wetland to converge into a single cell. All GIS analyses were performed with GRASS GIS 6.4 (GRASS Development Team, 2008) (see chapter 2).

Since irrigation flows from near irrigated areas were supposed to exert more influence on wetland hydrology (Castañeda and Herrero, 2008b) and their effect on larger wetlands should be considerably lower than in smaller ones, raw percentages of irrigated areas in the watersheds were weighted by landscape metrics, such as inverse-distance weights (King et al, 2005; Van Sickle and Johnson, 2008) and receiving wetland area. A wetland area relative percentage index (WARP) was thus used in order to assess and compare the hydrological pressures that irrigation at watershed scale exerts on different wetlands (see section 3.2.4 in chapter 3 and equation 4.1):

$$\mathrm{WARP}_{ILA} = \sqrt{\frac{\text{ILA (IDW) in watershed (\%)}}{\text{Wetland area (ha)}}} \tag{4.1}$$

Where ILA (IDW) refers to raw percentages of irrigated land areas in the watersheds, which were first inverse-distance weighted (IDW) . Maps of irrigated areas from each wetland watersheds in 2008 were obtained from previous studies, where land cover maps were produced by means of supervised classification of Landsat images at 25 m resolution (see chapter 2).

4.2.3. Vegetation field sampling

Across wetlands, a total of 1,843 georeferenced sampling plots (2 × 2 m) were surveyed between April and June 2009. The number of vegetation sampling plots within each wetland ranged from 30 to 550, systematically located at different intervals depending on wetland size and heterogeneity of plant communities. At each sampling plot, selected plant taxa cover was recorded (0-100%), together with upland species and bare soil cover. According to known qualitative ranges of tolerance to salinity and waterlogging, 21 representative perennial taxa were sampled: *Suaeda vera* Forssk. ex J.F. Gmel., *Atriplex halimus* L., *Atriplex glauca* L., *Arthrocnemum macrostachyum* (Moric.) Moris in Moris & Delponte, *Halimione portulacoides* (L.) Aellen, *Halocnemum strobilaceum* (Pall.) M. Bieb., *Sarcocornia fruticosa* (L.) A.J. Scott, *Suaeda pruinosa* Lange, *Scirpus holoschoenus* L., *Frankenia corymbosa* Desf., *Phragmites australis* (Cav.) Trin. ex Steud., *Lygeum spartum* Loefl. ex L., *Imperata cylindrica* (L.) Raeusch., *Juncus* sp. L., *Limonium caesium* (Girard.) Kuntze, *Limonium cossonianum* Kuntze, *Plantago crassifolia* Forssk., *Sarcocornia perennis* (Mill.) A.J. Scott, *Saccharum ravennae* (L.) Murray, *Tamarix boveana* Bunge and *Tamarix canariensis* Willd. Several sampling plots were disregarded due to the presence of infrastructures or other disturbances.

4.2.4. Ordination and classification analysis

Vegetation sampling data were grouped using ordination and classification analysis to identify vegetation clusters which might be related to the ones described by the European Habitat Directive. Taxa that are present in less than 1% of the sampling plots were discarded from the analysis. The final data matrix contained 13 taxa and 1,684 sampling plots. The taxa-abundance matrix was first transformed into a dissimilarity matrix using the $Bray - Curtis$ dissimilarity index. Hierarchical clustering was then performed and the resulting classification tree was cut in groups, representing plant communities. In order to determine significant indicator taxa of each plant community type indicator value analysis (IndVal) was applied (Roberts, 2010). Non Metric Multi Dimensional Scaling (MDS) was also conducted (Oksanen et al, 2011) to graphically represent and assess the fit of the ordination analysis. SIMPER procedure (Clarke, 1993) was used to determine average dissimilarities between vegetation clusters. Finally, analysis of similarities (ANOSIM) (Clarke, 1993) was carried out to test whether there were significant differences in plant composition between the obtained communities. All statistical analyses were conducted using R (R Core Team, 2013).

4.2.5. Wetland ecological condition index based on plant community composition

Supported by the knowledge on the ecology of each studied plant community type, and its sensitivity to hydrological related factors, a wetland condition index was sought, that could serve as a measure of semiarid Mediterranean saline wetland ecological condition status in relation to watershed agricultural hydrological pressures. In order to quantitatively analyze wetland plant communities composition and summarize them into a meaningful biological gradient, correspondence analysis (CA) was performed on the resulting wetland communities-abundance matrix (Benzécri, 1973; Husson et al, 2010). Hydro-ecological linkages between wetland plant communities composition (CA axes) and watershed hydrological condition (ILA WARP-IDW) were then established by means of regression analysis, and thus an index of wetland condition (WCI_{PC}) based on a plant community composition was developed.

4.2.6. Plant communities mapping

Plant communities of the studied wetlands were mapped by means of supervised classification of an image with a pixel size of 2 m, obtained with an airborne multispectral sensor in 2008 (DGPNB, 2008). The sensor used was a *DMC Z/I Intergraph* camera with four spectral bands, *i.e.* red (R, 590-675 nm), green (G, 500-650 nm), blue (B, 400-580 nm) and near infrared

(NIR, 675-850 nm). Normalised Difference Vegetation Index (NDVI; Rouse et al, 1973) was calculated and was included as an ancillary layer in classification analysis. Since the image corresponding to Playa de la Hita (CR20) wetland was damaged, plant communities from this site could not be mapped.

The classification procedure consisted of two steps. Firstly, unsupervised classification of 20 spectral clusters was performed to discriminate readily identifiable non vegetation cover classes. According to photointerpretation using an aerial photography obtained from the same flight (0.45 m pixel size), the obtained classes were identified as infrastructures, water bodies and open spaces of bare soil and were masked out from further classification analysis. Secondly, supervised classification of image was performed in order to discriminate wetland plant communities. For this purpose, georeferenced sampling plots (2×2 m) surveyed in the field were used as training and validation areas for image classification. Fifty percent of the pixels belonging to each plant community type were randomly selected and assigned to the training and validation maps, respectively. Although airborne image dated from June 2008, almost one year earlier than field sampling took place, plant species composition is known to be relatively constant in time in these ecosystems (Zedler et al, 1999) and almost no vegetation changes were expected during this period since no specific disturbances occurred. Random forest (Breiman, 2001) was the algorithm used for supervised classification. This nonparametric classifier grows a set of 500 decision trees using a different subset of train areas each time, allowing to vote for the most popular assigned class, and thus producing a significant increase in classification accuracy (Liaw and Wiener, 2002; Pal, 2005). Classification results were validated by means of the *Overall Accuracy* parameter and the Kappa coefficient (Foody, 2002). Producer's and user's accuracy parameters were also calculated for each plant community of interest across wetland sites (Congalton, 1991).

4.3. Results

4.3.1. Watershed hydrological conditions

The size of wetland watersheds ranged from 70 to 17,000 hectares. Percentages of irrigated areas among wetland watersheds ranged from 0% to 99% (Figure 4.3A). While Sombrerico watershed showed the highest percentage of irrigated area, they were absent in Cañada Brusca South and Rasall watersheds. When wetland area relative percentages (WARP) were calculated, larger wetlands showed relatively lower agricultural hydrological pressures than smaller ones, and as a result a gradient of watershed hydrological conditions was observed across wetlands (Figure 4.3B).

4.3.2. Plant communities characterization

The resulting hierarchical classification tree was cut in eight clusters and significant indicator taxa of each corresponding plant community type or cluster were identified using the IndVal analysis (Table 4.2). MDS representation of field sampling plots (stress of 0.18; Figure 4.4) and

Table 4.1: Study wetlands and respective watershed areas (ha).

Wetland	Wetland area	Watershed area
Salinas del Rasall	26.3	236
Saladar de Cañada Brusca South	3.8	69.5
Saladar de Cañada Brusca North	17.4	360
La Alcanara	199	6,508
Marina del Carmolí	314	16,923
Playa de la Hita	34.4	2,052.8
Saladar de Matalentisco	10.4	907.6
Saladar de Boquera de Tabala	36.9	5,819.2
Lopoyo	80	2,783
Ajauque	100	7,792
Derramadores	50	1,963
Sombrerico	3	141

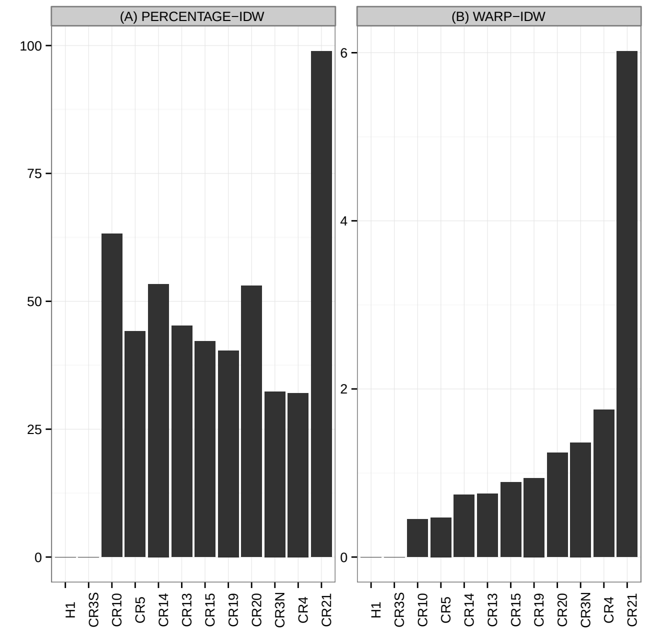

Figure 4.3: Irrigated agricultural land areas (ILA) percentages (inverse distance weighted: IDW) (A) and wetland area relative percentages (WARP-IDW) (B) in wetland watersheds in 2008. Wetland keys: **H1**: Rasall; **CR3S**: Cañada Brusca South; **CR10**: Carmoli; **CR5**: Alcanara; **CR14**: Ajauque; **CR13**: Lopoyo; **CR15**: Derramadores; **CR19**: Boquera de Tabala; **CR20**: Playa de la Hita; **CR3N**: Cañada Brusca North; **CR4**: Matalentisco; **CR21**: Sombrerico.

Table 4.2: Indicator species for each vegetation cluster after the IndVal Analysis. Associated probability value and corresponding code of the EU Habitats Directive is indicated.

Cluster name	Species	Indicator value	Probability	EU Habitat
SM-Ama	*A. macrostachyum*	0.786	0.001	1420
SM-Sfr	*S. fruticosa*	0.8148	0.001	1420
	Juncus sp.	0.0888	0.001	
SM-Hpo	*H. portulacoides*	0.7916	0.001	1420
SS	*F. corymbosa*	0.3618	0.001	1510
	L. spartum	0.3183	0.001	
	L. caesium	0.0961	0.001	
	L. cossonianum	0.0566	0.002	
Pau	*P. australis*	0.8596	0.001	–
Tca-Aha	*Tamarix sp.*	0.6983	0.001	92D0/1430
	A. halimus	0.4688	0.001	
Sve	*S. vera*	0.5408	0.001	1430
	A. glauca	0.0869	0.001	
UP	*Upland plants*	0.7747	0.001	–

ANOSIM analysis showed that vegetation clusters were clearly distinct ($R^2 = 0.83$; $P = 0.001$). Average dissimilarities between vegetation clusters were above 80% after the SIMPER analysis, except for two clusters dominated by *Suaeda vera* and several salt steppe taxa, respectively, which only reached 69%.

Most of the vegetation clusters obtained were consistent with plant communities described in the Habitat Directive. Typical salt-marsh communities (EU Habitat *1420* – Mediterranean and thermo-Atlantic halophilous scrubs) were differenciated in three groups, dominated by *Arthrocnemum macrostachyum* (SM-Ama), *Sarcocornia fruticosa* (SM-Sfr) and *Halimnione portulacoides* (SM-Hpo). Salt steppe community (SS) was evenly represented by several characteristic taxa and corresponded to EU Habitat *1510* (Mediterranean salt steppes – Limonietalia). A plant community was represented by *Phragmites australis* (Pau), which is not included in the Habitat Directive. *Tamarix* sp. (mainly *Tamarix canariensis*) and *Atriplex halimus* were grouped together in one plant community (Tca-Aha), representing dense Tamarisk forest patches surrounded by *A. halimus*. This type corresponds with a mixed community of EU habitats *92D0* (Southern riparian galleries and thickets) and *1430* (Halo-nitrophilous scrubs Pegano-Salsoletea). Another plant community was mainly represented by *Suaeda vera* (Sve), which also corresponded to a documented subtype of the EU Habitat *1430* composed by pioneering taxa in recently abandoned arable areas (MARM, 2009). Finally, a heterogeneous vegetation cluster was obtained with no specific indicator taxa, composed by several upland ruderal halo-nitrophilous plant species not specifically belonging to wetlands (UP).

4.3.3. Wetland ecological condition index based on plant community composition

The use of CA allowed us to summarize wetlands plant community composition into a single multivariate axis that represented a biological condition gradient, which can be related to environmental variables. Plant communities represented by *Phragmites australis*, *Tamarix canariensis* and *Atriplex halimus* were added up and pooled in a single functional plant community type, since all of them have been previously reported as invasive and related to hydrological perturbations (Burdick and Konisky, 2003; Chambers et al, 2003; Álvarez Rogel et al, 2006) (see Figure 4.5).

The first and second axes of the CA explained 30% and 28% of the variance, respectively. In relation to the first axis, characteristic wetland plant communities were grouped in the near zero and positive part of the axis (SM-Sfr, SM-Ama, SM-Hpo and SS), whose ecological tolerances are similar and known to occur under typical wetland abiotic ranges in terms of salinity, water table level and flooding regimes (Álvarez Rogel et al, 2000, 2007c). On the contrary, the lowest values of the first axis grouped plant communities previously related to hydrological perturbations (Tca-Aha-Pau), together with a community type which is typical from wetland areas altered by *in situ* factors (Sve) (Caballero, 1999). After linear regression analysis, the first axis of the CA was negatively correlated with the irrigated land areas (WARP-IDW) index in 2008 ($R^2 adj. = 0.33; P = 0.03$) (see Figure 4.6A). Sombrerico wetland was considered and outlier and therefore was excluded from the regression analysis since it represented extreme ILA (WARP-IDW) values.

A plant community based index of wetland condition (WCI$_{PC}$) was thus proposed based on CA axis 1. Relative abundances of plant communities scoring negatively in CA axis 1 were subtracted to abundances of positive scoring communities (see Equation 4.2).

$$WCI_{PC} = (RA_{SM} + RA_{SS}) - (RA_{Tca-Aha-Pau} + RA_{Sve}) \tag{4.2}$$

Where RA_{SM} stands for the sum of the relative abundances of salt marsh communities (SM-Sfr, SM-Ama and SM-Hpo), RA_{SS} stands for relative abundance of the salt steppe community, $RA_{Tca-Aha-Pau}$ stands for the sum of the relative abundances of the Tamarisk and reed beds dominated plant communities, and RA_{Sve} stands for the relative abundance of the *Suaeda vera* dominated plant community, expressed as parts per unit.

The values of the condition index for the study wetlands ranged from -1 to 1 (see Figure 4.7). Positive values represent wetland sites with good ecological status, while negative values stand for disturbed sites due to hydrological perturbation. Boquera de Tabala (CR19), Derramadores (CR15) and Cañada Brusca South (CR3S) wetlands scored over 0.5, representing very good

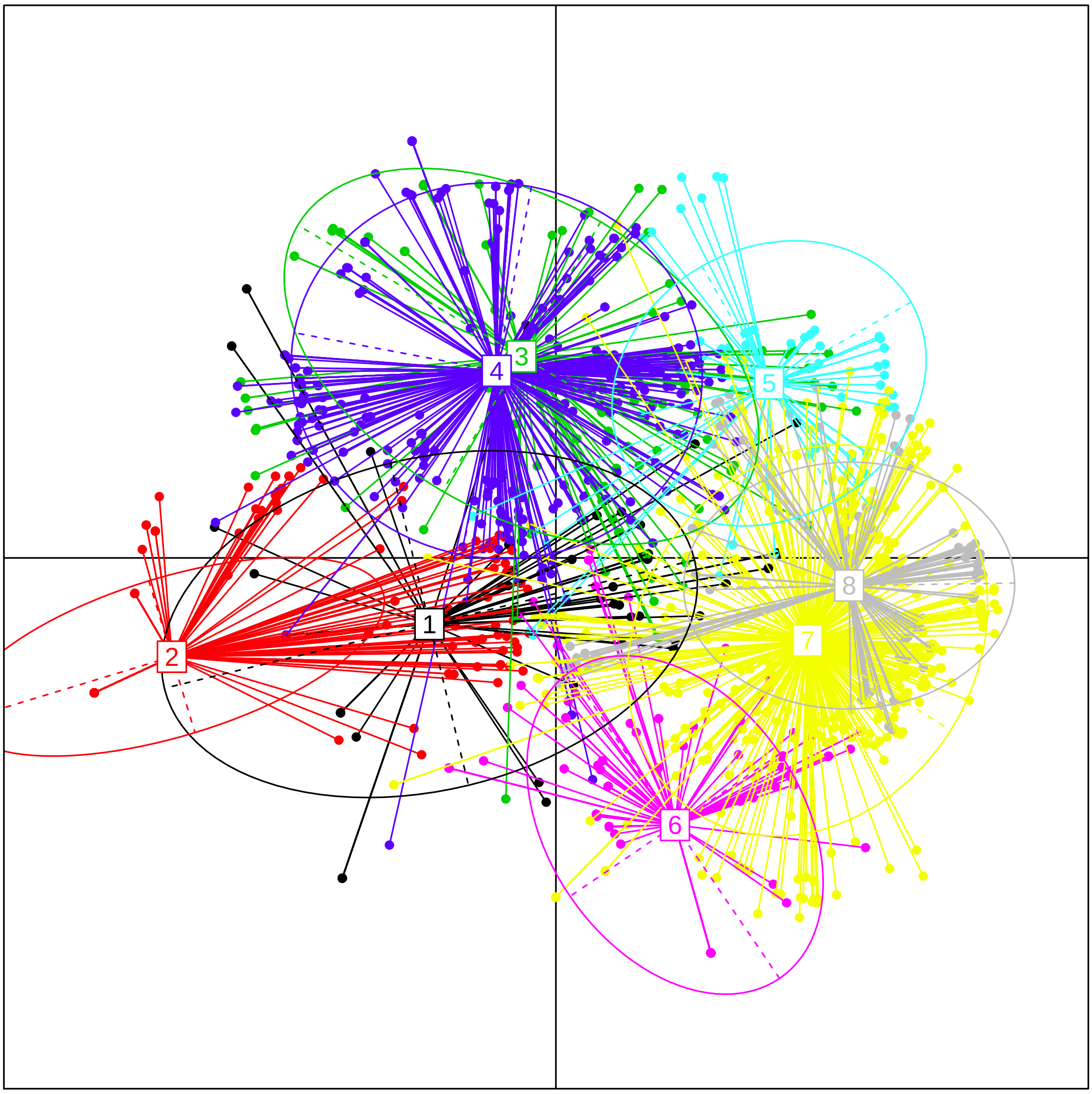

Figure 4.4: Non-metric multidimensional scale graph. Plant community keys: **1**: *Tamarix* sp. and *Atriplex halimus* mixed community (Tca-Aha); **2**: *Phragmites australis* dominated community (Pau); **3**: salt marsh dominated by *Arthrocnemum macrostachyum* (SM-Ama); **4**: salt marsh dominated by *Sarcocornia fruticosa* (SM-Sfr); **5**: salt marsh dominated by *Halimione portulacoides* (SM-Hpo); **6**: dominated by upland plants (UP); **7**: salt steppe (SS); **8**: *Suaeda vera* dominated community (Sve).

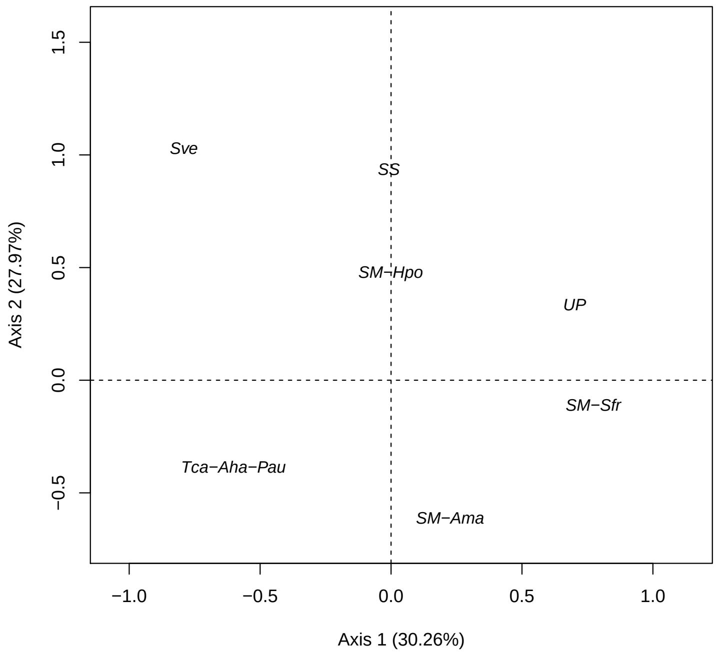

Figure 4.5: Correspondence analysis (CA) of wetland plant communities abundance. Plant community keys: **Tca-Aha-Pau**: *Tamarix* sp. + *Atriplex halimus* + *Phragmites australis*; **SM-Ama**: salt marsh dominated by *Arthrocnemum macrostachyum*; **SM-Sfr**: salt marsh dominated by *Sarcocornia fruticosa*; **SM-Hpo**: salt marsh dominated by *Halimione portulacoides*; **SS**: salt steppe; **Sve**: *Suaeda vera*; **UP**: upland plants.

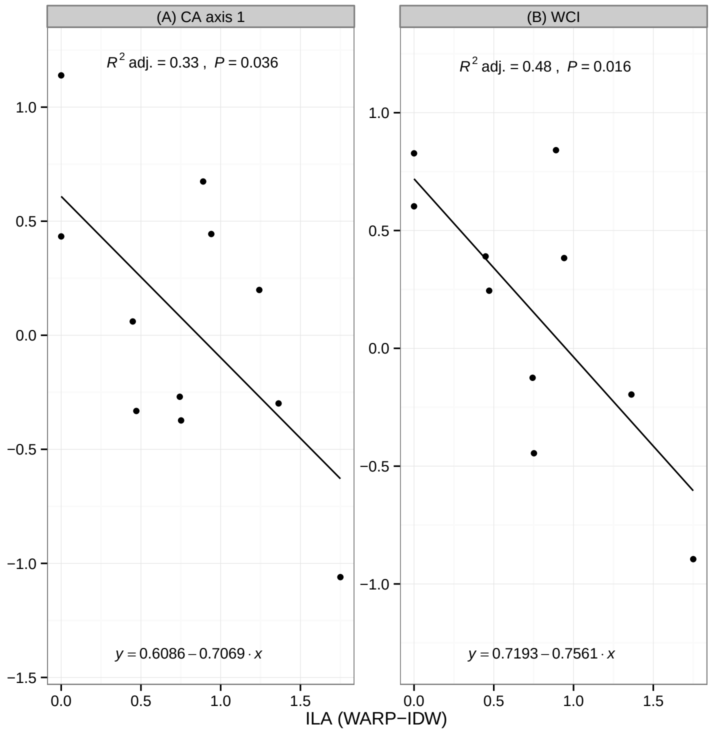

Figure 4.6: Relationship between watershed hydrological condition (ILA WARP-IDW) in 2008 and (A) correspondence analysis (CA) axis 1 and (B) plant community based index of wetland condition (WCI_{PC}) as the response variables.

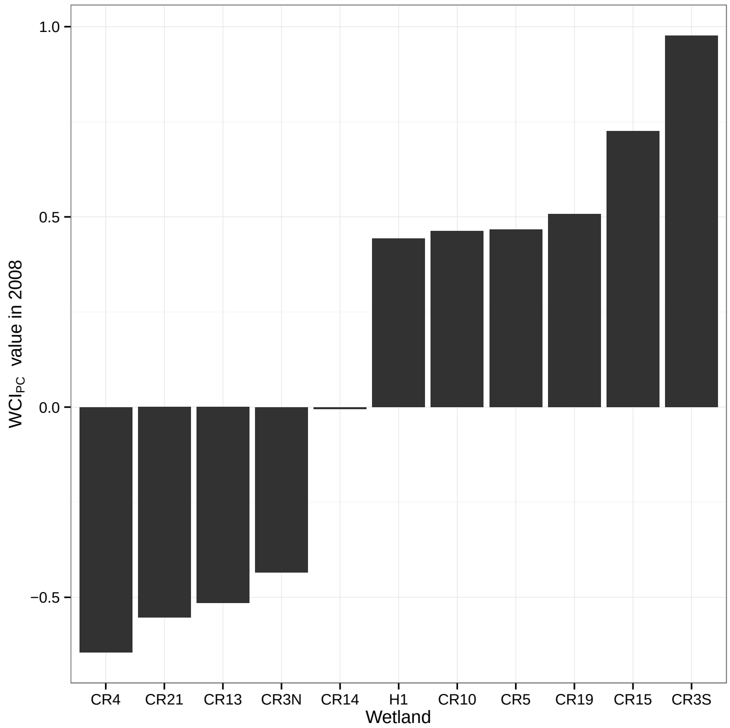

Figure 4.7: Values of the plant community based index of wetland condition (WCI_{PC}) in 2008. Wetland keys: **CR4**: Matalentisco; **CR21**: Sombrerico; **CR13**: Lopoyo; **CR3N**: Cañada Brusca North; **CR14**: Ajauque; **H1**: Rasall; **CR10**: Carmoli; **CR5**: Alcanara; **CR19**: Boquera de Tabala; **CR15**: Derramadores; **CR3S**: Cañada Brusca South.

condition sites, and Matalentisco (CR4), Sombrerico (CR21) and Lopoyo (CR13) wetlands showed condition values under -0.5, indicating very bad condition.

Interestingly, the proposed plant community based wetland condition index showed a stronger relationship with the irrigated land areas index (WARP-IDW; $R^2 adj. = 0.48; P = 0.02$; see Figure 4.6B) in 2008 than with the first axis of the correspondence analysis. There was no correlation between wetland condition and abundance of upland plants ($r = 0.17; P = 0.61$). The presence of this community type is probably related to an inaccurate or old wetland delimitation but no to hydrological pressures coming from the watershed, and therefore we did not include them in the condition index.

4.3.4. Plant community maps

A separate random forest classification model was performed for wetlands located in the South of Murcia province (CR3N, CR3S, CR4 and CR21) in order to improve classification accuracy at these sites. All other wetlands were classified using the same random forest model.

Resulting maps were validated for the plant communities included in the wetland condition index, *i.e.* Tca-Aha-Pau, SM (grouping SM-Ama, SM-Sfr and SM-Hpo), SS and Sve (Figure 4.8). Kappa and overall accuracy parameters varied among wetland sites (see Table 4.3).

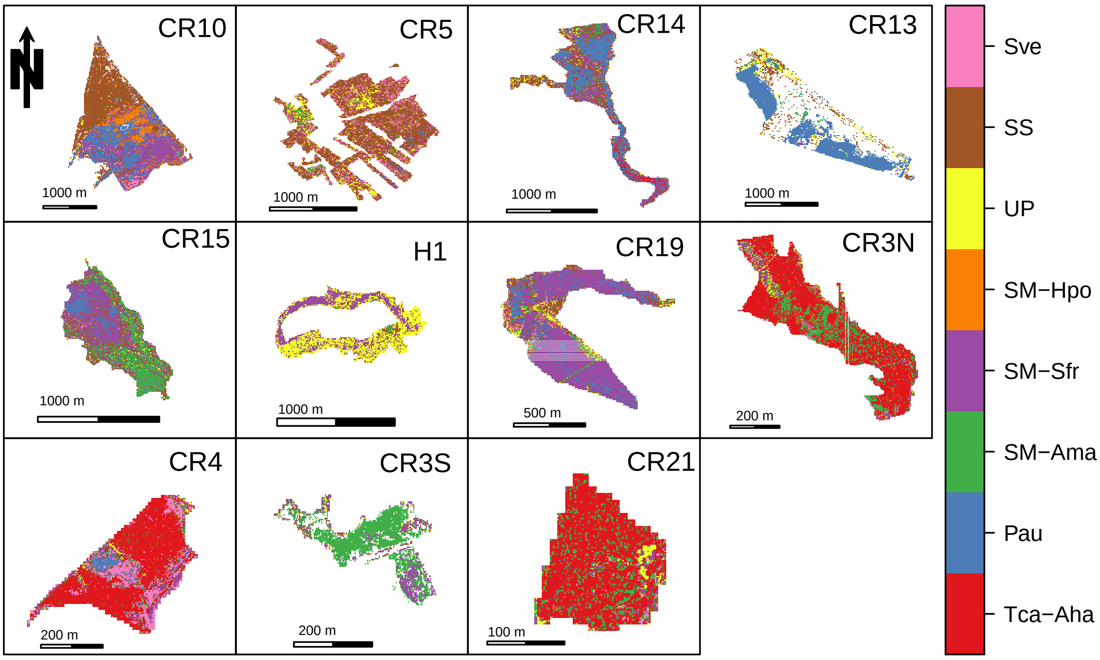

Figure 4.8: Wetland plant community maps. Wetland keys: **CR10**: Carmoli; **CR5**: Alcanara; **CR14**: Ajauque; **CR13**: Lopoyo; **CR15**: Derramadores; **H1**: Rasall; **CR19**: Boquera de Tabala; **CR3N**: Cañada Brusca North; **CR4**: Matalentisco; **CR3S**: Cañada Brusca South; **CR21**: Sombrerico. Plant community keys: **Sve**: *Suaeda vera* dominated; **SS**: salt steppe; **UP**: dominated by upland plants; **SM-Hpo**: salt marsh dominated by *Halimione portulacoides*; **SM-Sfr**: salt marsh dominated by *Sarcocornia fruticosa*; **SM-Ama**: salt marsh dominated by *Arthrocnemum macrostachyum*; **Pau**: *Phragmites australis* dominated; **Tca-Aha**: *Tamarix* sp. and *Atriplex halimus* mixed community.

Table 4.3: Overall accuracy and Kappa values for each wetland plant community map. Wetland sites are grouped by the two random forest (RF) models performed (Northern and Southern wetlands). Number of validation areas per wetland is indicated (N). Wetland keys: **CR10**: Carmoli; **CR13**: Lopoyo; **CR14**: Ajauque; **CR15**: Derramadores; **CR19**: Boquera de Tabala; **CR5**: Alcanara; **H1**: Rasall; **CR21**: Sombrerico; **CR3N**: Cañada Brusca North; **CR3S**: Cañada Brusca South; **CR4**: Matalentisco.

	Wetland	Overall Accuracy	Kappa	N
RF North	**CR10**	74%	0.62	243
	CR13	95%	0.78	42
	CR14	67%	0.55	85
	CR15	72%	0.26	72
	CR19	49%	0.10	51
	CR5	76%	0.51	96
	H1	94%	0.64	16
RF South	**CR21**	59%	0.20	17
	CR3N	52%	0.06	21
	CR3S	75%	0.33	12
	CR4	50%	0.20	22

Producer's and user's accuracy parameters calculated for each plant community of interest across wetland sites showed high accuracy results for most plant communities, except for *Sve*, which was mainly confused with SS and SM communities (see Table 4.4). Plant communities abundances obtained by means of remote sensing were significantly correlated with those estimated by means of field sampling across wetlands ($r > 0.9; P < 0.01$; Figure 4.9).

4.4. Discussion and conclusions

Our study comprises both basic and applied research by developing a rapid and effective technique to assess and monitor the ecological condition of semiarid Mediterranean wetlands. According to our results, the main objectives of the study were achieved: (1) plant community types were obtained in a systematic and reproducible way, (2) the proposed wetland condition index related wetland plant community composition and watershed hydrological pressures and (3) plant community maps obtained by means of remote sensing offered high accuracy and high spatial resolution in a wide range of wetland sizes. Our approach provides an effective procedure for using very high resolution airborne images of plant communities to evaluate the status of semiarid Mediterranean saline wetlands in relation to agricultural hydrological pressures at the catchment scale, thus serving as a tool for watershed management in order to prevent the actual wetland degradation rates (Zedler and Kercher, 2005; Dudgeon et al, 2006). The indicator wetland plant communities found in this study were coherent with

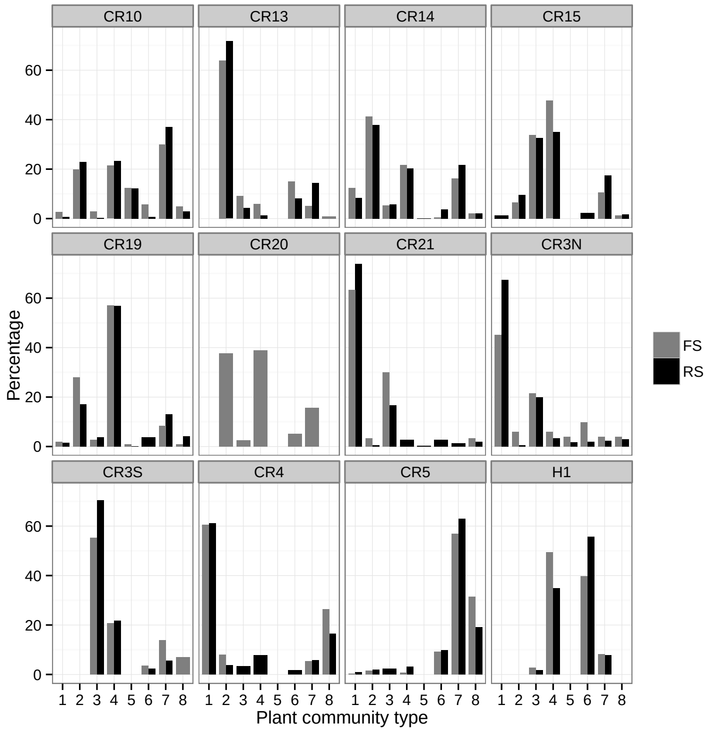

Figure 4.9: Plant communities abundance in the study wetlands according to field sampling (FS) and remote sensing (RS) data. Wetland keys: **CR10**: Carmoli; **CR13**: Lopoyo; **CR14**: Ajauque; **CR15**: Derramadores; **CR19**: Boquera de Tabala; **CR20**: Playa de la Hita; **CR21**: Sombrerico; **CR3N**: Cañada Brusca North; **CR3S**: Cañada Brusca South; **CR4**: Matalentisco; **CR5**: Alcanara; **H1**: Rasall. Plant community keys: **1**: *Tamarix* sp. and *Atriplex halimus* mixed community (Tca-Aha); **2**: *Phragmites australis* dominated community (Pau); **3**: salt marsh dominated by *Arthrocnemum macrostachyum* (SM-Ama); **4**: salt marsh dominated by *Sarcocornia fruticosa* (SM-Sfr); **5**: salt marsh dominated by *Halimione portulacoides* (SM-Hpo); **6**: dominated by upland plants (UP); **7**: salt steppe (SS); **8**: *Suaeda vera* dominated community (Sve).

Table 4.4: Error matrix calculated using all wetland plant community maps and validation sites. Plant community keys: Tca-Aha (*Tamarix* sp. + *Atriplex halimus*), Pau (*Phragmites australis*), SM (Salt marshes), SS (Salt steppe), Sve (*Suaeda vera*).

		Reference Data						
		Tca-Aha	Pau	SM	SS	Sve	Total	User's accuracy
Map data	Tca-Aha	**38**	5	11	4	3	61	62.30%
	Pau	2	**106**	24	5	4	141	75.18%
	SM	6	25	**175**	19	8	233	75.11%
	SS	3	8	29	**135**	9	184	73.37%
	Sve	4	7	9	10	**28**	58	48.28%
	Total	53	151	248	173	52	**677**	
	Producer's accuracy	71.70%	70.20%	70.56%	78.03%	53.85%		
					Overall Accuracy:	71%	Kappa:	0.34

indicator plant taxa obtained in previous studies for these type of wetland ecosystems and pressures (see chapter 3). Therefore, the study develops a cost-effective method for wetland regional assessments, thus overcoming the limitations of existing condition indices for semiarid Mediterranean saline wetland ecosystems (Ortega et al, 2004; Castañeda and Herrero, 2008b).

Wetlands contain diverse habitats, which are prone to generate different plant species and communities due to intrinsic natural variability (Innis et al, 2000), while acting as refugees for some native wetland plants against invasion by alien plant species (Schwartz and Jenkins, 2000). Hydrological perturbations represent a major threat to biodiversity in wetlands, that contain plant communities adapted to a specific range of soil salinity and flooding duration. Salt marsh taxa and communities can endure sporadic floodings and are very salt-tolerant (Pujol Fructuoso, 2002; Vicente et al, 2007), while salt steppe species are less frequent at flooded areas (Caballero, 1999; Álvarez Rogel et al, 2000). These ecosystems do no present substantial inter-annual variation in perennial taxa and plant community composition in absence of perturbations (see chapter 3). Furthermore, major differences in vegetation composition in semiarid Mediterranean wetlands are based on their relative abundances, rather than on taxonomic composition (Zedler et al, 1999), which is limited by abiotic conditions, such as soil salinity, anoxic conditions and waterlogging (Sanchez et al, 1998; Álvarez Rogel et al, 2007a). However, the invasion of semiarid Mediterranean saline wetlands by *Tamarix canariensis*, *Atriplex halimus* and *Phragmites australis* seems to have altered characteristic plant communities, resulting in the replacement of valuable halophytes by more generalist and opportunistic taxa, probably due to decreased salinities and perturbed flooding regimes. *Phragmites australis* usually forms monospecific stands in areas with regular water flows, reducing species diversity, and therefore indicating wetland degradation (Burdick and Konisky, 2003; Álvarez Rogel et al, 2007c; Tulbure et al, 2007), whereas *Atriplex halimus* in association with *Tamarisk canariensis* can form dense stands in areas with lower salinity conditions (MARM, 2009).

The fact that Matalentisco (CR4) and Boquera de Tabala (CR19) wetlands, both lacking of protection status, showed very contrasting ecological condition values due to waterhsed hydrological pressure reinforces the idea that watershed management plays an important role for wetland conservation (Figure 4.7; Wigand et al 1999; Turner et al 2003; Mack 2006). Another example is the contrasting plant community composition and condition index values of Cañada Brusca wetlands, whose North (CR3N) and South (CR3S) fragments clearly receive different surface hydrological influences (Figure 4.7). Positive index values can be considered as indicative of a good/fair ecological status, where natural plant communities dominate the wetland site, while negative values indicate dominance of invasive plants and wetland degradation due to strong hydrological pressures.

According with this and previous studies (Belluco et al, 2006; Rooney et al, 2012; Mwita et al, 2013), remote sensors have been shown to be suitable for the study and monitoring of wetland plant communities. After our results, random forest models showed to be valid only within the spatial extent in which they were developed. High spatial resolution played an important role when mapping small wetlands, reducing also the within-pixel heterogeneity, and therefore increasing their spectral separability (Belluco et al, 2006). However, plant communities are not discrete entities and their species composition may vary slightly across wetlands. Therefore, intraclass heterogeneity, as well as inaccuracies related to georeferenced sampling plots in the field, may have affected classification accuracy, which was comparable to results obtained on similar recent studies (Chen and Lin, 2013). Plant community represented by *Suaeda vera* (Sve) showed the lowest accuracy values (Table 4.4) probably due to the fact that it is associated to perturbed areas like abandoned agricultural areas, which might increase heterogeneity among sampling plots.

Watersheds and wetland sites in the Southern part of Murcia province were among the most perturbed after the application of the watershed condition (WARP-IDW; Figure 4.3) and the wetland condition (WCI_{PC}; Figure 4.7) indices, respectively. Moreover, image classification of plant communities at these wetland sites showed generally less accuracy than in northern wetlands (Table 4.3). This might reflect the fact that their plant communities composition deviates from the characteristic ones due to the invasion of opportunistic taxa, which make them more heterogeneous and difficult to discriminate by means of remote sensing.

In this study, we only disposed of one single image and therefore we lacked of phenological information of vegetation, which would surely improve the discrimination of plant communities. Due to the increasing public availability of remotely sensed data, together with the use of free and open source software (Steiniger and Hay, 2009), the proposed wetland monitoring and condition assessment method can be easily extended and further developed in the Mediterranean region (Herrero and Castañeda, 2009; MacKay et al, 2009), preferably using several images representing different seasons (Xie et al, 2008) for image classification analysis. For this purpose, the use of other advanced remote sensing sources like present and forthcoming satellite sensors is advised, *e.g.* SPOT, Landsat-8 and Sentinel 2 missions (Drusch et al, 2012; Irons et al, 2012), since they offer multi-temporal images of medium and high spatial and spectral resolution, respectively.

5 A spatio-dynamic R model and library of plant communities responses to hydrological pressures in a semiarid Mediterranean wetland

Abstract

Semiarid Mediterranean saline wetlands are semi-terrestrial ecosystems, which yearly undergo dry periods of several months, and shelter a rich, endemic and sensitive biota. In the last decades, the expansion of agricultural irrigated areas in semiarid Mediterranean catchments has led to altered inputs of water and nutrients to lowland wetlands. Hydrological alterations have affected characteristic plant communities, resulting in the replacement of valuable halophilic salt marsh and salt steppe plant communities by more generalist and opportunistic taxa like reed beds. A spatio-dynamic model and library were developed using R that aimed to explain the spatial distribution of three characteristic wetland plant communities in a semiarid Mediterranean wetland site in response to hydrological pressures from the catchment. Wetland plant communities and watershed irrigated agricultural areas were mapped by means of remote sensing at several dates between 1984 and 2008 and partly used as forcing inputs and validation data. A dynamic model was initially developed using Stella software and then converted into R by means of the StellaR software. Spatial dimension was added including neighborhood and spatial flows algorithms representing plant communities dispersion. The conversion between plant communities was caused by the increase in water inflows from the watershed, mediated by spatial parameters, such as the distance to ephemeral rivers and the flow accumulation map within the wetland site. Results of the model were in agreement with remote sensing data, showing that in 2008 salt steppe had lost a half of its original area, whereas salt marsh and reed beds experienced an important expansion process. The model developed in this study is available online as an R library, including all necessary input data sets and maps and documentation to run it. Free and open source software and online code sharing repositories are proposed as modelling tools for future research. The model library offers a flexible tool that suits the needs of both advanced and less skilled modelers.

5.1. Introduction

Semiarid Mediterranean saline wetlands are unique and endangered ecosystems which shelter high biodiversity. These are highly complex ecosystems in which the distribution of plant communities is mainly according to spatial environmental gradients of water availability and salinity (Álvarez Rogel et al, 2001, 2006). During last decades, land use changes in Murcia province and the associated hydrological changes at watershed scale have caused a rise in water tables of aquifers and have increased the levels of groundwater, flooding periods and soil water content in the wetlands (Álvarez Rogel et al, 2007c). Characteristic plant communities are negatively affected by these pressures to the wetland, especially salt steppe, while opportunistic invasive species are favored, such as reed beds (Chambers et al, 1999; Burdick and Konisky, 2003; Maheu-giroux and Blois, 2005). These changes are relevant from a biodiversity and conservation perspective since salt steppe is considered of Priority interest according to the European Habitat Directive (Council of Europe, 1992).

The failure to perceive that wetlands are not standalone elements in the landscape and to understand or express the complexity of spatial relationships among hydrology and wetland vegetation, has led to an extensive loss of the most characteristics wetland habitats during the last decades (Turner et al, 2000; Cools et al, 2013). Modelling the physical environment of wetlands is therefore essential for assessing the relationships among pressures and species responses in these threatened ecosystems (Zhou et al, 2008). Spatio-dynamic modelling has served as a tool to assess the effects of land use changes on wetland ecosystems and to substantiate and improve wetland restoration measures (Jørgensen and Bendoricchio, 2001; Hattermann et al, 2006). Moreover, spatial modelling has become an important tool for plant ecology and biodiversity studies (Turner et al, 1995; Moloney and Jeltsch, 2008; Gardner and Engelhardt, 2008). Furthermore, linking remote sensing, field studies and spatio-dynamic modelling can help overcoming some typical limitations of ecological studies, such as the lack of historical data or the difficulty of studying species interactions and their relationships with environmental variables and pressures both in space and time (Damgaard, 2003; Perry and Millington, 2008; Chen et al, 2011).

In the last decades there have been important advances in spatio-dynamic modelling, leading to a diversity of modelling environments, such as SME (Spatial Modelling Environment; Maxwell and Costanza 1997), MOHID (Braunschweig et al, 2004), Simile (Muetzelfeldt and Massheder, 2003), SimuMap (Pullar, 2004), Tarsier (Watson and Rahman, 2004) and TerraME (de Senna Carneiro et al, 2013). However, some of them remain unavailable for the research community and others represent commercial solutions, which limits their use and development. Furthermore, some key issues remains still open, such as the compatibility between models, the lack of reuseability and transparency and the targeted end-users or developers communities (Argent et al, 2006; Jørgensen, 2008). To propose a solution, we identified several requirements for modelling tools including: (1) they should allow interoperability between models; (2) represent adequately documented software tools that can be useful for non-programmers and (3) be flexible enough, so that advanced users can fully understand the role of each component and adapt them to case-specific requirements. The challenge is to develop modelling tools as general and flexible as possible to suit the needs of both advanced and less skilled modelers (Voinov et al, 2004; Jakeman et al, 2006). Trying to address these challenges, we have used R (R Core Team, 2013) as a modelling environment for fitting the above identified needs with an application to the spatio-dynamic ecological modelling of semiarid Mediterranean saline wetlands.

R is an advanced statistical computing system that is freely available for most computing platforms and can be used for modelling. Among other advantages, R has dynamic modelling, GIS and advanced graphics capabilities by means of multiple libraries (Petzoldt and Rinke, 2007; Pebesma, 2012; Hijmans et al, 2013), offers the possibility of developing new functions and can easily connect to other free and open source GIS, such as GRASS GIS (GRASS Development Team, 2008), SAGA GIS (Development Team, 2010) or Quantum GIS (Development Team, 2009). Besides, R is used extensively in elementary teaching, for student projects and by researchers including those in companies and it is supported by a worldwide community of users (Ripley, 2001). Modelers needs to be aware of the best practices and modelling techniques to improve their own approaches. It is essential that researchers make their models available and adopt common modelling tools and methodologies based on free and open source software tools (FOSS), which are accessible and can be easily improved by their community of users (Turner et al, 1995; Argent, 2004).

The model developed in this study addresses three plant communities in the Marina del Carmolí wetland in terms of spatio-dynamic interactions and responses to agricultural hydrological pressures from the catchment area over a 24 years period. The specific aims of the study were: 1) To develop an ecological model of a semiarid Mediterranean wetland site; 2) to build a user-friendly, online and documented version of a spatio-dynamic modelling library using R and 3) to test the wetland model by means of empirical and remote sensing data for the period 1984 – 2008.

5.2. Study area

A semiarid Mediterranean saline coastal wetland was selected, which comprises 314 hectares (figure 5.1). The Marina del Carmolí wetland is located in the Campo de Cartagena lowland coastal plain, associated to the internal shore of the Mar Menor coastal lagoon, which comprises 12,700 ha (Conesa, 1990; Conesa and Jiménez-Cárceles, 2007). The Campo de Cartagena area is under the influence of a very arid climate with a mean annual temperature higher than 18°C and mean annual rainfall under 300 mm (Conesa, 1990). This site is a Special Protection Area for Birds, a Site of Community Importance and a Special Protection Area for the Mediterranean).

The wetland mainly comprises salt steppe, salt marsh and reed beds areas, which are distributed according to the availability of water and salinity. Salt steppe is located in areas with low water availability and high salinities; reed beds (*Phragmites australis*) are in areas with high water content and low salinity, whereas salt marsh occupies areas with intermediate water

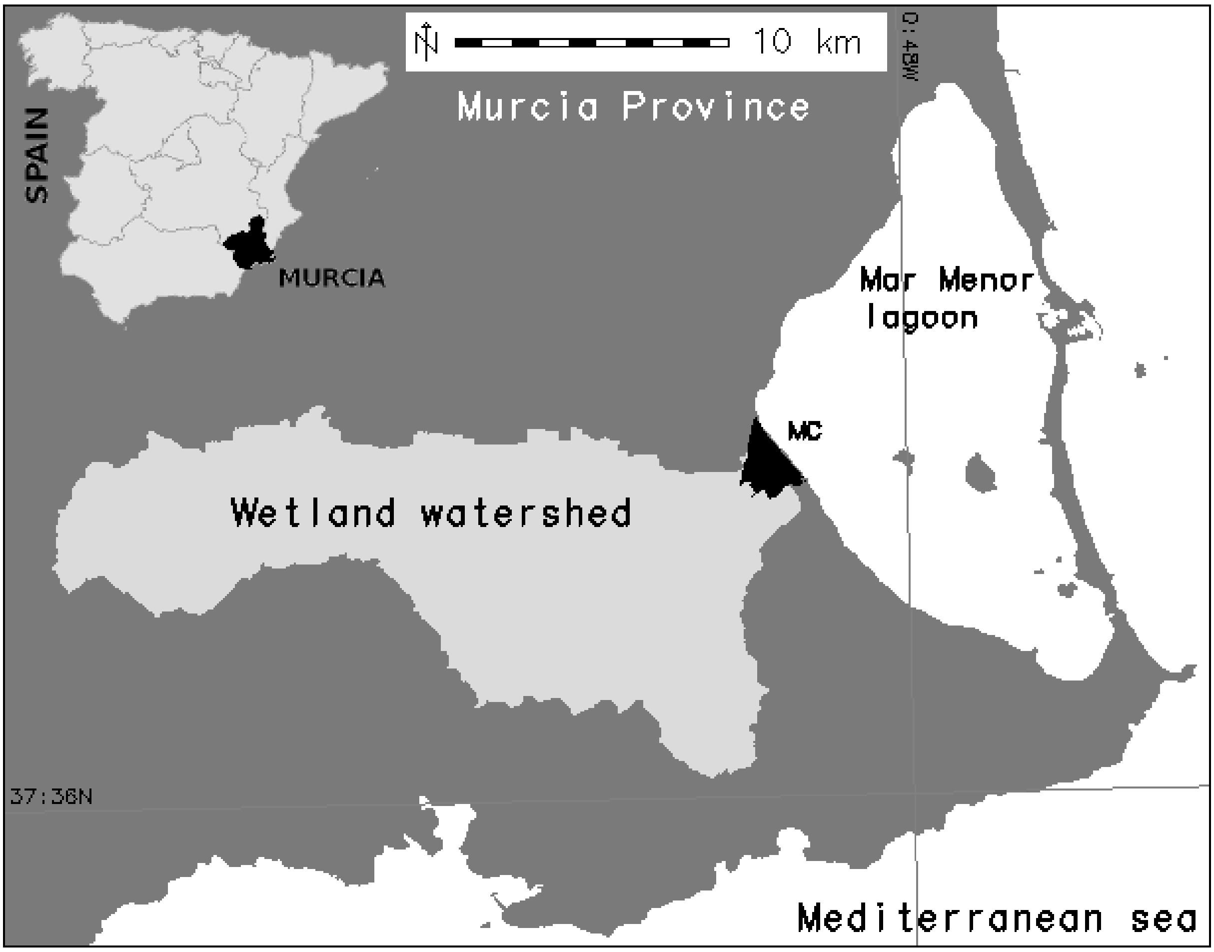

Figure 5.1: Location map of the Marina del Carmolí wetland (MC) and its watershed area in Murcia province (SE Spain).

content and higher salinity (Álvarez Rogel et al, 2006, 2007c; MARM, 2009). Salt steppe is considered of Priority interest according to the European Habitat Directive (Council of Europe, 1992), salt marsh is of Community interest and reed beds are not included in the Directive.

5.3. Model variables, parameters and assumptions

The three above mentioned plant community types (*i.e.* salt steppe, salt marsh and reed beds) and the bare soil were the state variables of the model and were represented as raster maps with a pixel size of 25 m. Initial maps of salt steppe, salt marshes and bare soil in the wetland were established according to a map obtained by means of remote sensing map in year 1984 (Carreño et al, 2008). Since reed bed stands were not dense enough to be mapped by remote sensors at that early stage, we did not know its initial location in the wetland and therefore it was assumed to be potentially present in all wetland pixels with an initial value of one unit in each pixel. As an invasive and clonal species whose rapid expansion occurs by extending its rhizomes in all directions, this seemed an ecologically meaningful assumption (Bart and Hartman, 2003; Engloner, 2009).

Conversion among plant communities was caused by the drainage water input to the wetland and mediated by spatial environmental variables influencing water availability and growth. The environmental factors considered in the model were: (1) the potential surface flow accumulation over the wetland area, as a proxy of waterlogging (spatial parameter); (2) the distance to the ephemeral rivers crossing the wetland (spatial parameter); and (3) the irrigated agricultural hydrological pressures coming from the watershed over time (non spatial forcing input). The model assumes only increasing or no water inputs, thus accounting only for the conversion of drier and more saline plant communities to more humid or less saline ones, *i.e.* salt steppe and bare soil (initially present) into salt marsh, and for the conversion of all of them into reed beds. The water availability and growth of reed beds was mainly influenced by the proximity to the ephemeral rivers, whereas the salt marsh community was influenced by the potential flow accumulation in combination with the average distance to both ephemeral rivers (Pennings and Callaway, 1992; Álvarez Rogel et al, 2001, 2006, 2007c; Carreño et al, 2008; Moffett et al, 2010).

In order to assess the agricultural hydrological pressures coming from the watershed over time, first the watershed area draining to the wetland was delineated from a 10 m raster DEM using GRASS GIS 6.4 (GRASS Development Team, 2008) (see chapter 2). Historical land cover maps for years 1987, 1996, 2000 and 2008 were obtained by means of remote sensing techniques (Carreño et al, 2011) (see chapter 2). The percentage of irrigated agricultural areas in the watershed was then calculated and a converted into a Wetland Area Relative Percentage index (WARP; see chapters 3 and 4), which was included in the model as a forcing input representing the agricultural hydrological pressure on the wetland over time. WARP values for intermediate years were obtained by means of linear interpolation (figure 5.2).

The surface flow accumulation map over the wetland area was derived from a 10 m DEM by means of watershed modelling operation using GRASS GIS. Maps of distance of each pixel to two ephemeral rivers crossing the wetland were produced by means of map algebra operations using a digitized river network. Resulting maps of flow accumulation and distance to ephemeral channels were scaled to a zero to one range in order to use them as normalized parameters of potential water availability for the expansion of the reed beds and salt marsh communities (figure 5.3).

Spatial neighborhood algorithms were developed and included in the model in order to allow the salt marsh community to disperse to the surrounding pixels. The dispersion of salt marsh was limited to neighboring pixels containing bare soil or salt steppe. Figure 5.4 shows a diagram of the relationships among the different variables of the model.

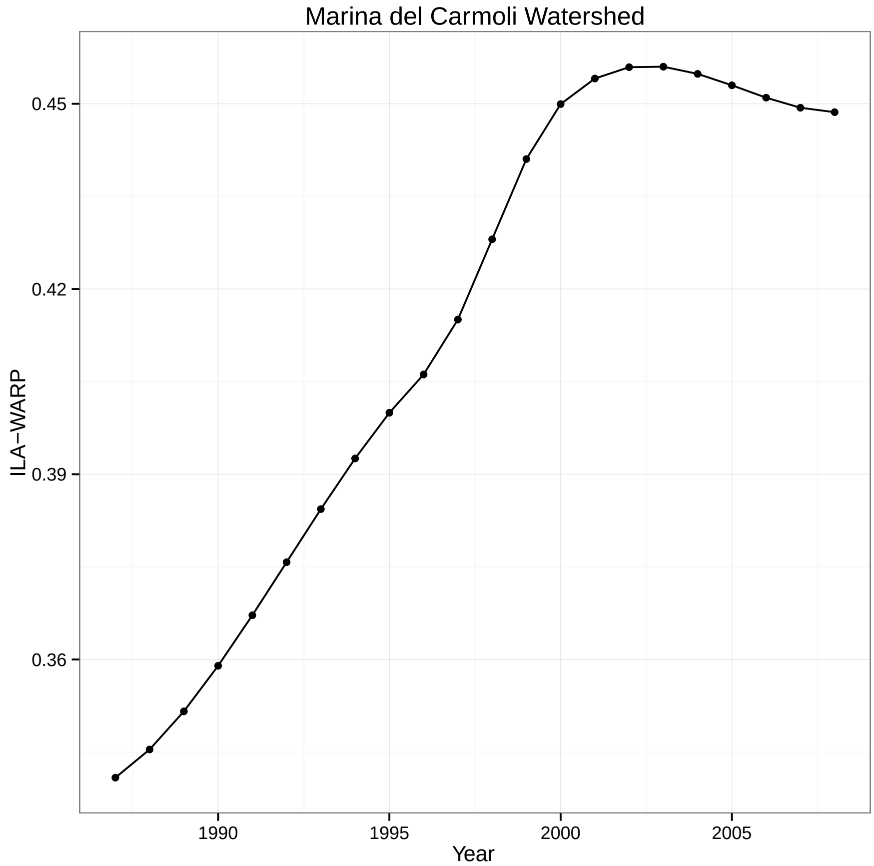

Figure 5.2: Wetland Area Relative Percentage (WARP) index of irrigated agriculture during the study period in the Marina del Carmolí wetland watershed according to remote sensing data.

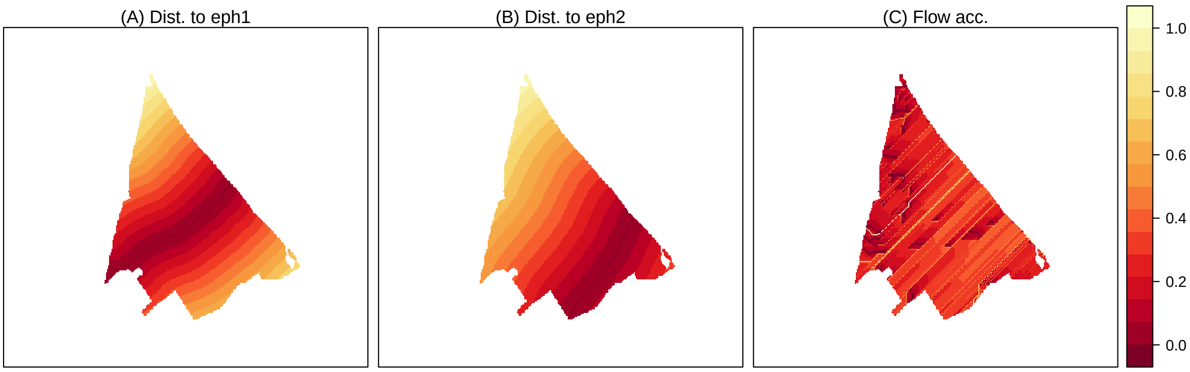

Figure 5.3: Environmental spatial variables included in the model: (A) distance map to ephemeral river 1; (B) distance map to ephemeral river 2; (C) Flow accumulation map. All variables are on a relative 0–1 scale.

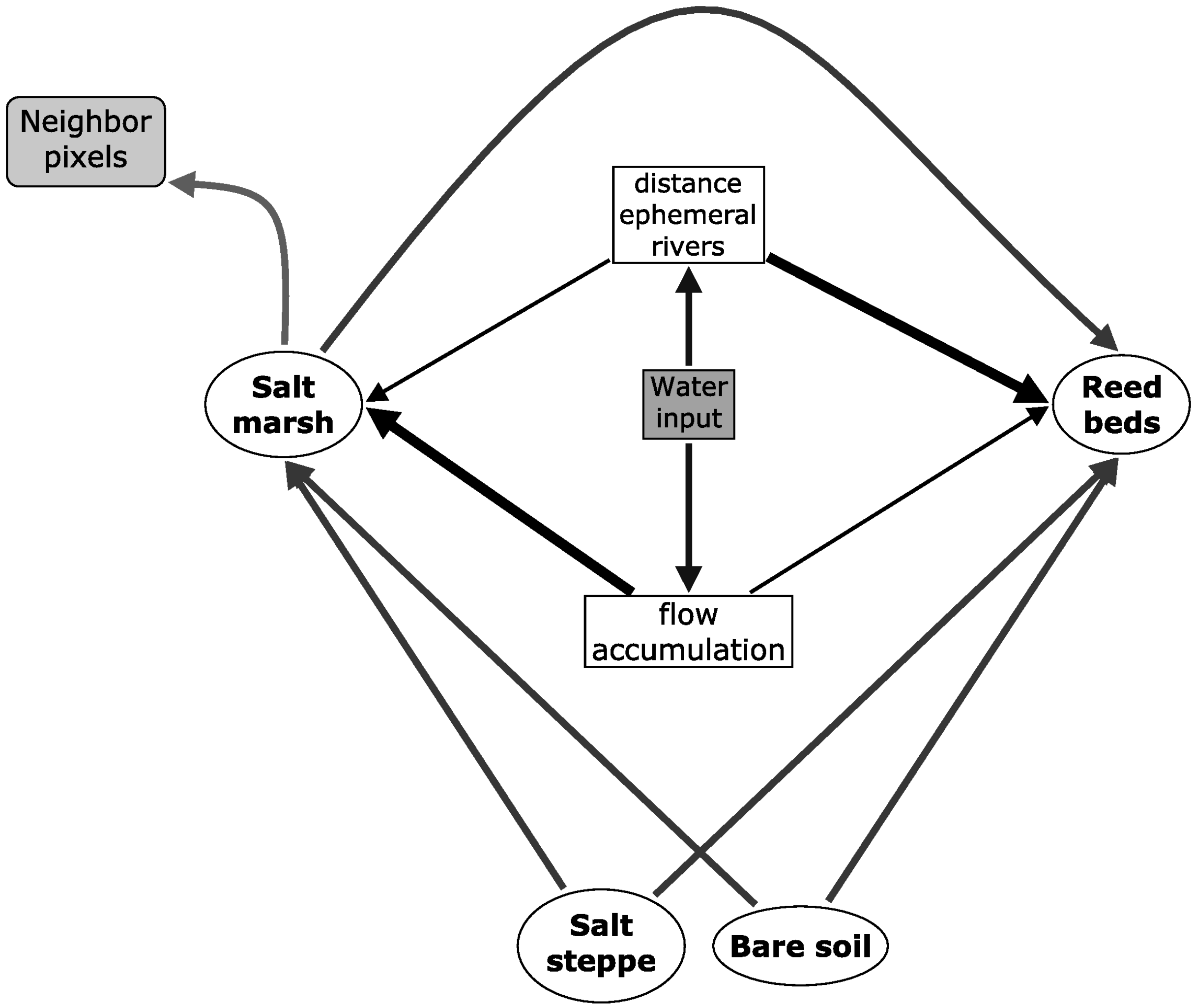

Figure 5.4: Model conceptual diagram. Thicker lines refer to stronger dependencies between parameters and state variables.

Model variables and parameters were established using a deterministic approach based on the knowledge of the ecological tolerance of the plant communities and environmental variables, such as the maximum potential growth rates of reed beds and salt marsh communities, the effective water availability for each plant community and the water dependent conversion coefficients between plant communities. Table 5.1 shows the list of model variables, parameters and equations implemented in the model. Figure 5.5 summarizes all model parameters and equations related to the conversion between plant community types.

Wetland plant community maps from years 1992, 1995, 1997, 2001 and 2008, obtained by means of remote sensing, with an overall accuracy ranging from 74% to 89% (Carreño et al, 2008; Esteve et al, 2008; Martínez-López et al, 2012), were used as independent validation data for assessing the results of the model. Besides, the model was also tested under a no drainage water input scenario.

Table 5.1: Model variables, parameters and equations.

Name	Short name	Range/Value/Description
State variables		
Salt steppe	ss	0–25
Salt marsh	sm	0–25
Reed beds	rb	0–25
Bare soil	bs	0–25
Spatial parameters		
Flow accumulation + (1 - norm. aver. dist. to ephemeral rivers)	fa_ave_ephs	0–2
Distance to ephemeral river 1 and 2	eph1 and eph2	0–1
Non spatial parameters		
Potential growth rate of reed beds	pgr_rb	0.005
Potential growth rate of salt marsh	pgr_sm	0.2
Forcing input		
ILA (WARP-IDW)	ilawarp	$\geqq 0$
Equations		
Potential water availability for reed beds	pwarb	$eph1^5 + eph2^5$
Potential water availability for salt marsh	pwasm	$fa\_ave\_ephs^4$
Effective water availability for reed beds	ewarb	$(1 + ilawarp) * pwarb$
Effective water availability for salt marsh	ewasm	$(1 + ilawarp) * pwasm$
Water dependent conversion coefficient from salt steppe to salt marsh	wdcc_ss_sm	$ilawarp/2$
Water dependent conversion coefficient from bare soil to salt marsh	wdcc_bs_sm	$ilawarp/0.2$
Water dependent conversion coefficient from salt steppe to reed beds	wdcc_ss_rb	$ilawarp/0.45$
Water dependent conversion coefficient from bare soil to reed beds	wdcc_bs_rb	$ilawarp/0.1$
Water dependent conversion coefficient from salt marsh to reed beds	wdcc_sm_rb	$ilawarp/3$
Actual conversion rate from salt marsh to reed beds	acr_sm_rb	$ewarb * wdcc\_sm\_rb * pgr\_rb$
Actual conversion rate from salt steppe to reed beds	acr_ss_rb	$ewarb * wdcc\_ss\_rb * pgr\_rb$
Actual conversion rate from bare soil to salt marsh	acr_bs_sm	$ewasm * wdcc\_bs\_sm * pgr\_sm$
Actual conversion rate from salt steppe to salt marsh	acr_ss_sm	$ewasm * wdcc\_ss\_sm * pgr\_sm$
Actual conversion rate from bare soil to reed beds	acr_bs_rb	$ewarb * wdcc\_bs\_rb * pgr\_rb$
Total conversion from salt steppe to reed beds	tc_ss_rb	$rb * acr\_ss\_rb * ss * (1 - (rb/25))$
Total conversion from salt steppe to salt marsh	tc_ss_sm	$sm * acr\_ss\_sm * ss * (1 - (sm/25))$
Total conversion from salt marsh to reed beds	tc_sm_rb	$rb * acr\_sm\_rb * sm * (1 - (rb/25))$
Total conversion from bare soil to salt marsh	tc_bs_sm	$sm * acr\_bs\_sm * bs * (1 - (sm/25))$
Total conversion from bare soil to reed beds	tc_bs_rb	$rb * acr\_bs\_rb * bs * (1 - (rb/25))$
Salt marsh growth	smg	$tc\_ss\_sm + tc\_bs\_sm - tc\_sm\_rb$
Reed beds growth	rbg	$tc\_ss\_rb + tc\_sm\_rb + tc\_bs\_rb$
Salt steppe decrease	ssd	$-tc\_ss\_rb - tc\_ss\_sm$
Bare soil decrease	bsd	$-tc\_bs\_sm - tc\_bs\_rb$

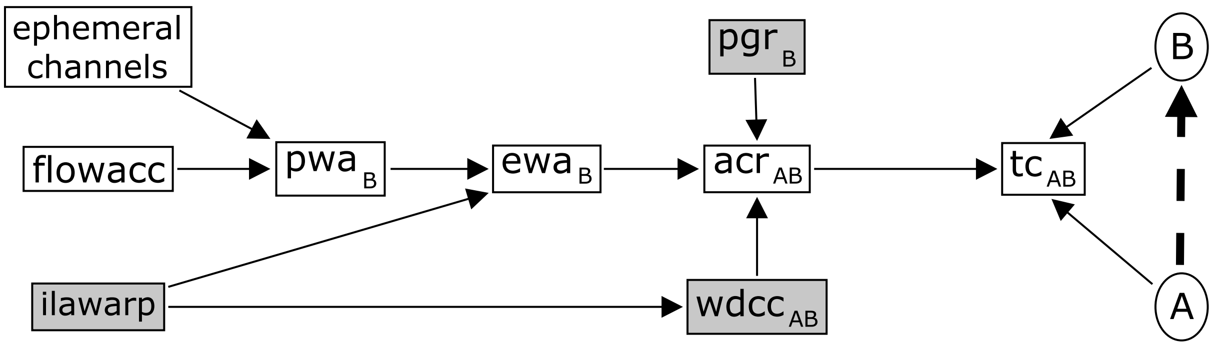

Figure 5.5: Example of the conversion among plant communities showing the relationships between the main model parameters and variables. A and B subindices refer to the conversion from plant community type A to B. Shaded squares represent non spatial variables and parameters, and oval shaped variables refer to state variables. Variables and parameters legend: 'flowacc': flow accumulation; 'ephemeral rivers': average distance to ephemmeral rivers; 'ilawarp': hydrological pressure index; 'pwa': potential water availability for a community type; 'wdcc': water dependent conversion coefficient from one community type to another; 'ewa': effective water availability for a community type; 'pgr': potential growth rate of a community type; 'acr': actual conversion rate from one community type to another; 'tc': total conversion from one community type to another; 'A': Plant community type A; 'B': Plant community type B. For more information see table 5.1.

Table 5.2: Area (ha) occupied by each plant community after remote sensing (RS) and model results (MOD) for the years with initial and validation data.

	Salt steppe		**Salt marsh**		**Reed beds**	
	RS	**MOD**	**RS**	**MOD**	**RS**	**MOD**
1984	179		2		0	
1992	149	155	72	54	0	15
1995	108	136	127	83	9	41
1997	104	124	99	96	32	53
2001	117	109	81	114	63	63
2008	90	90	121	137	74	65

5.4. Model development and execution

The model was developed starting from a basic dynamic model using Stella (ISEE Systems Inc., 2013) which was translated into R code using the StellaR software (Naimi and Voinov, 2012). Different available R libraries were used for model development, mainly the 'deSolve' package for solving differential equations (Soetaert et al, 2010) and the 'raster' package for the analysis of spatial data (Hijmans, 2013). The model code was finally wrapped as an R function that was then solved by numerical integration using the *ode.2D* function and the Euler integration method (Soetaert et al, 2010). State variables and spatial parameters were defined in the model as matrices in order to comply with the requirements of this function. Time step was set to 0.25 and the 'Euler' method was selected as the integration algorithm. The model output is a matrix in which each row contains data from a specific time step and the columns correspond to all wetland pixels arranged by the different state variables. The total execution time of the model was approximately 60 seconds using a regular desktop computer with 4GB of RAM. For validation purposes, the resulting wetland maps of abundances for each state variable were summarized into a categorical map in which each pixel was assigned the state variable with the highest abundance.

5.5. Simulation results and validation

The model was able to spatially simulate the abundance of the plant communities of interest in a timely manner during the study period, accounting for (1) the differential effects of the environmental parameters selected on each plant community; (2) the potential invasive effect of reed beds; and (3) the effect of dispersion of the salt marsh community. According to the values simulated by the model and those obtained by means of remote sensing (table 5.2), between 1984 and 2008 the area of salt steppe was reduced to the half, being mainly replaced by salt marsh, which became the dominant community. On the contrary, reed beds, practically absent in 1984, occupied 65 hectares in 2008, after a significant expansion process since 1995.

Table 5.3: Error and similarity measures of the resulting areas occupied by each plant community across study years after remote sensing and spatio-dynamic modelling. 'r' corresponds to the Pearson correlation coefficient ($P < 0.05$), 'NMRSE' to the normalized root mean squared error and 'EF' to the efficiency factor. Data from 5 years were compared.

Plant community	r	NRMSE (%)	EF
Salt steppe	0.91	44	0.77
Salt marsh	0.84	54	0.65
Reed beds	0.88	53	0.65

Table 5.4: Overall accuracy of plant communities maps resulting from the model compared to the maps obtained by means of remote sensing. All pixels in the wetland were compared (approx. 4,000 pixels).

Year	Overall Accuracy
1992	71%
1995	55%
1997	58%
2001	54%
2008	61%

Results of the model showed high accuracy values in relation to the data obtained by means of remote sensing, which were used only for validation. Figure 5.6 and table 5.3 show the graphical representation and the performance measures of the model simulated versus the observed values regarding the area occupied by each plant community, respectively (Jachner et al, 2007; Bennett et al, 2013). The comparison between the model simulated wetland plant communities maps and those obtained by means of remote sensing for each validation date showed overall accuracy values up to 71% (Figure 5.7 and Table 5.4). Moreover, the model showed almost no change in plant communities abundance or dispersion during the study period when tested under no drainage water inputs, in accordance with the model assumptions.

5.6. Model library

The

model developed in this study is available online as an R library, including all necessary input data sets and maps and documentation to run it (https://github.com/javimarlop/spdynmod). The idea was to set a model library on an online collaborative environment where both end-users and advanced R users and modelers were able to run it, contribute to its development, as well as adapt it to their specific needs. GitHub (www.github.com) was selected as a free online

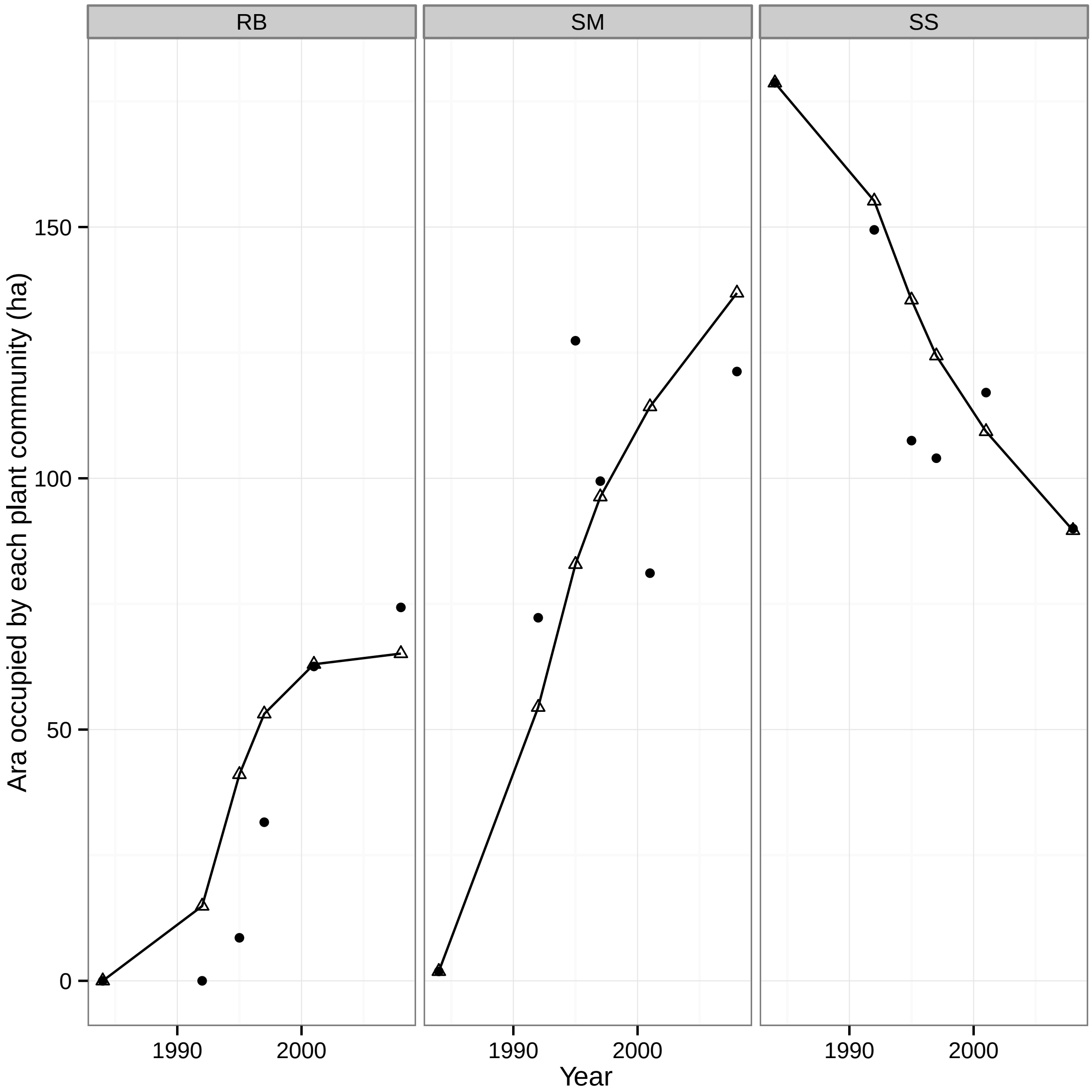

Figure 5.6: Comparison of the area (ha) occupied by each plant community after remote sensing and model simulated values. Dots represent values obtained by means of remote sensing, while model simulated values are represented by triangles linked through lines for more clarity. Legend: reed beds (RB), salt marsh (SM) and salt steppe (SS) plant communities.

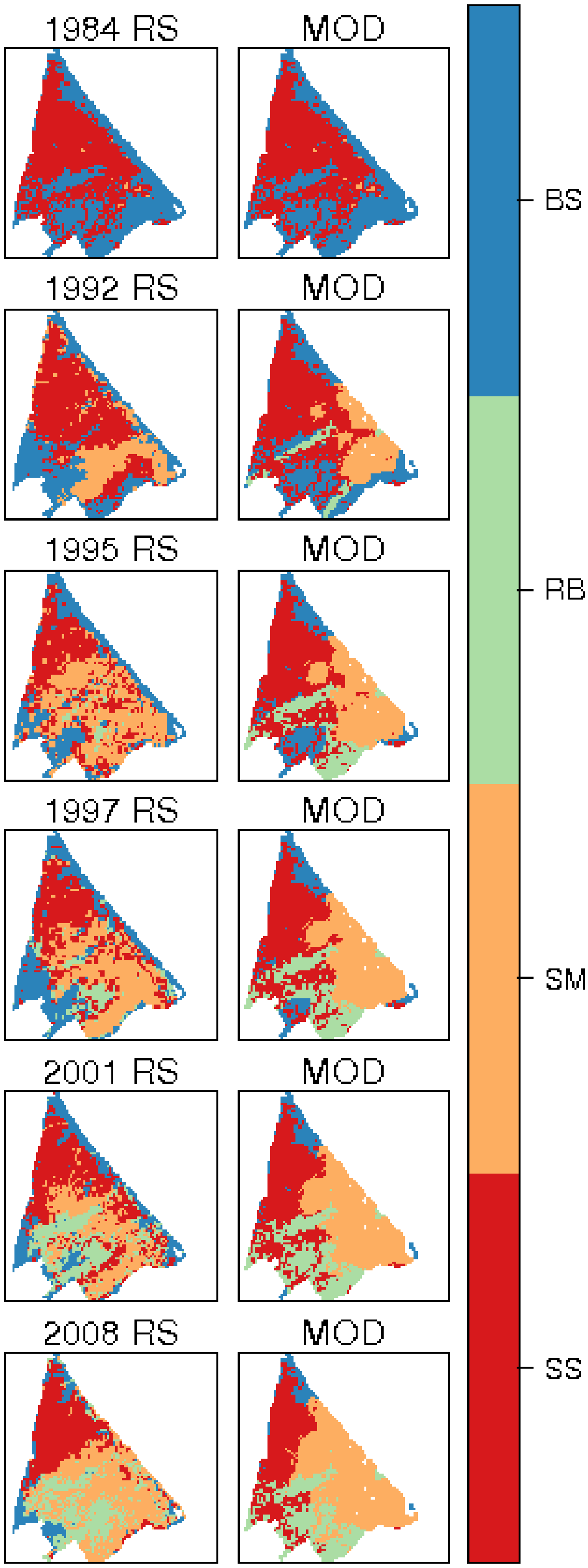

Figure 5.7: Wetland plant community maps over the study period after remote sensing (RS) and model simulated values (MOD).

repository hosting platform used for code sharing, from which the model library can be directly installed from the command line in R by means of the 'devtools' library (Wickham and Chang, 2013). Several functions are included in the initial version of the library that allow to run the model, display and animate the simulation results, and change some optional parameters, such as the potential growth rate of the salt marsh and reed beds communities. Besides, there is the possibility to establish random abundances and spatial configurations of the initial state variables, except for the reed bed community, in order to perform some model testing. Each function is documented according to the standard R documentation guidelines, which includes a description of the function, the input and output parameters, and some optional explanatory notes.

Further development of this library are constantly ongoing by the authors, yet the contribution from other advanced R users is expected to extend the library with other applications in a more generic context, such as model testing in similar wetlands, the inclusion of different environmental variables and ecosystems, model sensitivity analyses, etc. Although the model library can be used independently from the original Stella model, the StellaR script remains as a tool that allow modelers to connect both pieces of software for building and running new models in order to include the spatial dimension to their existing Stella models.

5.7. Discussion and conclusions

Overall, this study demonstrates the suitability of open source software, such as R, for developing models that are open to the community and which are able to integrate complex historical environmental and biological variables over time and space. The model served as a research tool for testing plant community interactions and the relationships between plant communities and environmental variables in space and time. The model is easy to interpret, mainly based on deterministic empirical data, openly accessible and easily reproducible. Furthermore, by means of the R library, the model could be extended by other researches in order to include other conceptual wetland models of soil-plant relationships (González-Alcaraz et al, 2013a), as well as broader topics, such as the distribution, morphology and habitats of saline wetlands (Castañeda et al, 2013). Besides, the use of the model library by environmental agencies as a management tool would contribute to the efficient protection and monitoring of semiarid saline Mediterranean wetlands by providing a scientific assessment on the responses of their plant communities to watershed agricultural pressures, as well as a tool to study wetland restoration measures. The model has at present some limitations, such as lack of sensitivity analysis or the possibility to study a different wetland site but further developments are foreseen in future versions of the model, after receiving some feedback from users testing.

Wetlands are complex ecosystems exhibiting strong spatial heterogeneity, which makes them hard to approach (Zhou et al, 2008). Therefore, including the spatial dimension of the main model parameters and state variables allowed us to better understand the distribution of each plant community type in response to the selected pressures. Results of the model clearly show that conventional protection and conservation strategies usually do not take into account the close dependency of wetlands on the dynamics and management outside the protected area and that this may interfere on the protection and conservation goals of protected wetlands (Turner et al, 1995). The relative changes between salt steppe, salt marsh and reed beds can be probably explained by the interaction between soil moisture and conductivity gradients (González-Alcaraz et al, 2013a). The initial increase of water inflows from the basin resulted in increased soil moisture and higher salinity, which favored the expansion of salt marsh at the expense of salt steppe. At a later stage, around 1995, the increased water inputs reduced water salinity and allowed the expansion of reed beds. The net loss of salt steppe is very important since it is the habitat of Mar Menor wetlands with the highest interest from the point of view of the European Habitat Directive.

The model library developed in this study can be easily linked with other modelling and GIS tools, such as python, PCRaster and TGRASS, in order to expand its visualization capabilities or to build integrated models (Karssenberg et al, 2007; Schmitz et al, 2013; Gebbert and Pebesma, 2014). Besides, R gives access to classic and state-of-the-art statistical methods and it also possesses several libraries for obtaining directly biological and environmental data from the web, such as GBIF data (GBIF, 2013), which makes it suitable for web modelling approaches (Dubois et al, 2013). Furthermore, R also offers specific libraries, such as 'shiny' (RStudio Inc., 2013), that add graphical interface capabilities, which could be integrated in any existing model. Although the model developed in this study is not computationally time consuming, R also offers several parallel computation libraries, allowing the implementation of large scale ecological models.

Between graphical interfaces and programming languages there are a wide variety of modelling tools available, that can help convert conceptual ideas into models (Costanza and Voinov, 2001). We agree that visual modelling environments that do not involve programming are easier to work with for high level users, especially for well developed models and non advanced users. However, command-line environments allow translating any concept into working computer code, and students and researchers are forced to think about what they are doing when they use them (Ripley, 2001). Besides, free and open source software offers a promising approach for developing standard modelling approaches, protocols and tools to collaborate in biodiversity research among researchers, students, decision makers and citizens in general (Steiniger and Hay, 2009; Rocchini and Neteler, 2012). The adoption of an open source community approach is especially crucial to make progress on important conservation challenges, like the implementation of large-scale ecological models to meet the Aichi Biodiversity Targets (COP 10 Decision X/2, 2010; Voinov et al, 2010).

The use of online hosting repositories and platforms for collaborative model development, documentation and exchange should be encouraged, similarly to past initiatives, such as ECOBAS (Hoch et al, 1998; Benz et al, 2001). In this regard, there are several free online repository hosting services that include additional functionalities to code sharing, such as version control, bug tracking, release maintenance, web and wiki services, which can be used as collaborative environments for model development, such as GitHub, BitBucket and SourceForge (Wilson et al, 2013). In fact, open scientific research is based on an open-source community through the use of such collaborative environments that facilitate model peer review, replication of results and delivery of results to the public, especially for publicly funded science projects (Voinov et al, 2010; Poisot et al, 2013).

General conclusions

1. The results of this thesis have shown how the ecological status of these semiarid Mediterranean saline wetlands was negatively affected by the increase in irrigated agricultural areas during last decades, which stress the importance of monitoring wetland watersheds for their conservation.

2. An enhanced free and open source procedure for watershed modelling in coastal plain areas was specifically developed for wetlands, which allowed to delineate wetlands watershed with high accuracy, especially for large wetlands.

3. An enhanced free and open source supervised image classification methodology with a high efficiency in terms of accuracy achieved and time consumption was used and further developed, which allowed to generate historical land cover maps for monitoring wetland watersheds.

4. Results showed that the frequency of wetland characteristic plant taxa were significantly related to irrigated agricultural areas at watershed scale during the 20-year study period.

5. The index based on plant taxa composition and frequency developed to relate irrigated agricultural areas to wetland ecological status at watershed scale resulted in an appropriate tool for wetland monitoring and assessment by means of fieldwork sampling.

6. The watershed hydrological condition (Wetland Area Relative Percentage; WARP) index developed allowed to successfully relate the percentage of irrigated agriculture and natural areas with the ecological status of wetlands at watershed scale, irrespective of their size, and its use is proposed for future studies.

7. The establishment of plant communities in the study wetlands by means of fieldwork sampling and ordination and classification analysis combined with their mapping by means of airborne remote remote sensors allowed the remotely-sensed characterization of wetlands at 2 meters spatial resolution with high accuracy.

8. An index for assessing wetland ecological status based on plant communities composition was developed in relation to watershed hydrological condition and proposed as a tool for wetland monitoring and assessment by means of remote sensing. Thus, remote sensing ultimately allowed the effective assessment and monitoring of semiarid Mediterranean saline wetland ecological status.

9. The spatio-dynamic model developed and tested for the Marina del Carmolí wetland successfully explained the expansion of reed beds and salt marsh communities in response to increasing watershed hydrological pressures during the study period, and how plant communities abundance and zonation respond to factors such as the distance to ephemeral rivers and potential flow accumulation, which ultimately influence plant water availability.

10. Free and open source software environments, such as R, allowed the development of a spatio-dynamic environmental model that can be applied to the study of semiarid Mediterranean saline wetlands. In this regard, the 'spdynmod' model library is freely available and can be used and further developed by the community.

Bibliography

Adam E, Mutanga O, Rugege D (2010) Multispectral and hyperspectral remote sensing for identification and mapping of wetland vegetation: a review. Wetlands Ecology and Management 18:281–296

Águila Guillén M, Hernández Guirao A, Núñez Herrero MA (2007) Guías de itinerarios didácticos por los espacios naturales de la Región de Murcia : humedales. Cuadernos CREA 7, URL http://hdl.handle.net/11162/1245

Aguirre-Gutiérrez J, Seijmonsbergen AC, Duivenvoorden JF (2012) Optimizing land cover classification accuracy for change detection, a combined pixel-based and object-based approach in a mountainous area in Mexico. Applied Geography 34(0):29 – 37

Ahmad S, Simonovic S (2004) Spatial System Dynamics: New Approach for Simulation of Water Resources Systems. Journal of Computing in Civil Engineering 18(4):331–340

Alexandridis TK, Zalidis GC, Silleos NG (2008) Mapping irrigated area in Mediterranean basins using low cost satellite Earth Observation. Computers and Electronics in Agriculture 64(2):93–103

Allan JD (2004) Landscapes and riverscapes: The influence of land use on stream ecosystems. Annual Review of Ecology Evolution and Systematics 35:257–284

Álvarez Rogel J, Alcaraz Ariza F, Ortiz Silla R (2000) Soil salinity and moisture gradients and plant zonation in Mediterranean salt marshes of Southeast Spain. Wetlands 20:357–372

Álvarez Rogel J, Ortiz Silla R, Alacaraz Ariza F (2001) Edaphic characterization and soil ionic composition influencing plant zonation in a semiarid Mediterranean salt marsh. Geoderma 99(1-2):81–98

Álvarez Rogel J, Martínez-Sánchez J, Blázquez L, Semitiel C (2006) A conceptual model of salt marsh plant distribution in coastal dunes of Southeastern Spain. Wetlands 26(3):703–717

Álvarez Rogel J, Carrasco L, Marín C, Martínez-Sánchez J (2007a) Soils of a dune coastal salt marsh system in relation to groundwater level, micro-topography and vegetation under a semiarid Mediterranean climate in SE Spain. CATENA 69(2):111 – 121

Álvarez Rogel J, Jiménez-Cárceles FJ, Egea-Nicolás C (2007b) Phosphorus retention in a coastal salt marsh in SE Spain. Science of The Total Environment 378(1–2):71 – 74

Álvarez Rogel J, Jiménez-Cárceles FJ, Roca, R Ortiz MJ (2007c) Changes in soils and vegetation in a Mediterranean coastal salt marsh impacted by human activities. Estuarine Coastal and Shelf Science 73(3-4):510–526

Antrop M (1993) The transformation of the Mediterranean landscapes: an experience of 25 years of observations. Landscape and Urban Planning 24(1-4):3 – 13

Aranzabal ID, Schmitz MF, Aguilera P, Pineda FD (2008) Modelling of landscape changes derived from the dynamics of socio-ecological systems: A case of study in a semiarid Mediterranean landscape. Ecological Indicators 8(5):672 – 685

Argent R, Voinov A, Maxwell T, Cuddy S, Rahman J, Seaton S, Vertessy R, Braddock R (2006) Comparing modelling frameworks - A workshop approach. Environmental Modelling & Software 21(7):895 – 910

Argent RM (2004) An overview of model integration for environmental applications - components, frameworks and semantics. Environmental Modelling & Software 19(3):219 – 234

Aspinall R, Pearson D (2000) Integrated geographical assessment of environmental condition in water catchments: Linking landscape ecology, environmental modelling and GIS. Journal of Environmental Management 59(4):299 – 319

Austoni M, Giordani G, Viaroli P, Zaldívar J (2007) Application of specific exergy to macrophytes as an integrated index of environmental quality for coastal lagoons. Ecological Indicators 7(2):229 – 238

Baker C, Lawrence R, Montagne C, Patten D (2006a) Mapping wetlands and riparian areas using Landsat ETM+ imagery and decision-tree-based models. Wetlands 26:465–474

Baker ME, Weller DE, Jordan TE (2006b) Comparison of automated watershed delineations : Effects on land cover areas, percentages, and relationships to nutrient discharge. Photogrammetric Engineering and Remote Sensing 72(2):159 – 168

Bannari A, Morin D, Bonn F, Huete AR (1995) A review of vegetation indices. Remote Sensing Reviews 13(1):95–120

Barberá G, Calvo JF, Esteve MA, Hernández V, Robledano F (1990) Importance of small man-made wetlands for breeding waders in South-eastern Spain. Wader St Group Bull 60:24–26

Barbier EB (2011) Wetlands as natural assets. Hydrological Sciences Journal 56(8):1360–1373

Bart D, Hartman J (2003) The role of large rhizome dispersal and low salinity windows in the establishment of common reed, *Phragmites australis*, in salt marshes: New links to human activities. Estuaries 26:436–443

Basset A, Sabetta L, Carrada GC (2006) Conservation of transitional water ecosystems in the Mediterranean area: bridging basic ecological research and theories with requirements of application. Aquatic Conservation: Marine and Freshwater Ecosystems 16(5):439–440

Basset A, Barbone E, Borja A, Brucet S, Pinna M, Quintana X, Reizopoulou S, Rosati I, Simboura N (2012) A benthic macroinvertebrate size spectra index for implementing the Water Framework Directive in coastal lagoons in Mediterranean and Black Sea ecoregions. Ecological Indicators 12(1):72 – 83

Belluco E, Camuffo M, Ferrari S, Modenese L, Silvestri S, Marani A, Marani M (2006) Mapping salt-marsh vegetation by multispectral and hyperspectral remote sensing. Remote sensing of environment 105(1):54–67

Bennett ND, Croke BF, Guariso G, Guillaume JH, Hamilton SH, Jakeman AJ, Marsili-Libelli S, Newham LT, Norton JP, Perrin C, Pierce SA, Robson B, Seppelt R, Voinov AA, Fath BD, Andreassian V (2013) Characterising performance of environmental models. Environmental Modelling & Software 40(0):1 – 20

Benz J, Hoch R, Legović T (2001) ECOBAS - modelling and documentation. Ecological Modelling 138(1 - 3):3 – 15

Benzécri JP (1973) L'analyse des données. Dunod, Paris-Bruxelles-Montréal

Blaschke T (2010) Object based image analysis for remote sensing. ISPRS Journal of Photogrammetry and Remote Sensing 65(1):2–16

Boix D, Gascón S, Sala J, Martinoy M, Gifre J, Quintana XD (2005) A new index of water quality assessment in Mediterranean wetlands based on crustacean and insect assemblages: the case of Catalunya (NE Iberian peninsula). Aquatic Conservation: Marine and Freshwater Ecosystems 15(6):635–651

Bradley BA, Mustard JF (2005) Identifying land cover variability distinct from land cover change: Cheatgrass in the great basin. Remote Sensing of Environment 94(2):204–213

Braunschweig F, Chambel P, Fernandes L, Pina P, Neves R (2004) The object-oriented design of the integrated modelling system MOHID. In: Computational Methods in Water Resources International Conference, Chapel Hill, North Carolina, USA

Breiman L (2001) Random forests. Machine Learning 45:5–32

Brinson MM, Malvárez AI (2002) Temperate freshwater wetlands: types, status, and threats. Environmental Conservation 29(02):115–133

Brown KM, Foody GM, Atkinson PM (2009) Estimating perpixel thematic uncertainty in remote sensing classifications. International Journal of Remote Sensing 30(1):209–229

Buchanan GM, Nelson A, Mayaux P, Hartley A, Donald PF (2009) Delivering a global, terrestrial, biodiversity observation system through remote sensing. Conservation Biology 23(2):499–502

Burdick D, Konisky R (2003) Determinants of expansion for *Phragmites australis*, common reed, in natural and impacted coastal marshes. Estuaries and Coasts 26:407–416

Caballero JM (1999) Vegetación de los saladares del Guadalentín (SE Ibérico) : estructura y dinámica. PhD thesis, Universidad de Murcia

Caballero JM, Calvo JF, Esteve MA, Nicolás E, Robledano F (1996) Saladares del Guadalentín, Spain. In: Management of Mediterranean Wetlands, Ministerio de Medio Ambiente, Madrid, pp 109–127

Caçador I, Neto J, Duarte B, Barroso D, Pinto M, Marques J (2013) Development of an Angiosperm Quality Assessment Index (AQuA-Index) for ecological quality evaluation of Portuguese water bodies—A multi-metric approach. Ecological Indicators 25:141 – 148

Callow JN, Niel KPV, Boggs GS (2007) How does modifying a DEM to reflect known hydrology affect subsequent terrain analysis? Journal of Hydrology 332(1-2):30 – 39

Calvo JF, Esteve Selma MÁ, Caballero JM (2002) Comparación entre grupos de vegetación obtenidos mediante métodos multivariantes y sintáxones en la descripción de comunidades vegetales halófilas. Anales de Biología (24):3–20

Calvo JF, Esteve Selma MA, Pardo MT, Caballero JM (2003) Relaciones especies-área en los saladares del Guadalentín (SE Ibérico, España) e implicaciones para la conservación, restauración y gestión. Anales de Biología (25):91–102

Cāmara G, Souza R, Freitas U, Garrido J (1996) SPRING: Integrating remote sensing and GIS by object-oriented data modelling. Computers & Graphics 20(3):395–403

Carignan V, Villard MA (2002) Selecting indicator species to monitor ecological integrity: A review. Environmental Monitoring and Assessment 78:45–61

Carreño MF, Martínez-López J, Palazón JA, Conzález JC, Esteve MA, Martínez J (2011) Aplicación de nuevas técnicas para la clasificación supervisada de imágenes Landsat en la determinación de usos del suelo: por píxel y por mancha. Mapping Interactivo marzo-abril:68–73

Carreño MF, Pardo MT, Esteve MÁ, Martínez J (2007) Dinámica de hábitats de los humedales litorales de la laguna del Mar Menor (Murcia, SE de España) asociada a los cambios en el régimen hidrológico de su cuenca drenante. Anales de Biología 29:13–22

Carreño MF, Esteve MA, Martínez J, Palazón JA, Pardo MT (2008) Habitat changes in coastal wetlands associated to hydrological changes in the watershed. Estuarine Coastal and Shelf Science 77(3):475–483

Castañeda C, Herrero J (2008a) Assessing the degradation of saline wetlands in an arid agricultural region in Spain. CATENA 72(2):205 – 213

Castañeda C, Herrero J (2008b) Measuring the condition of saline wetlands threatened by agricultural intensification. Pedosphere 18(1):11–23

Castañeda C, Herrero J, Conesa JA (2013) Distribution, morphology and habitats of saline wetlands: a case study from Monegros, Spain. Geologica Acta 11

Chambers R, Osgood D, Bart D, Montalto F (2003) *Phragmites australis* invasion and expansion in tidal wetlands: Interactions among salinity, sulfide, and hydrology. Estuaries 26:398–406

Chambers RM, Meyerson LA, Saltonstall K (1999) Expansion of *Phragmites australis* into tidal wetlands of North America. Aquatic Botany 64:261–273

Chander G, Markham BL, Helder DL (2009) Summary of current radiometric calibration coefficients for Landsat MSS, TM, ETM+, and EO-1 ALI sensors. Remote Sensing of Environment 113(5):893 – 903

Chavez PS (1996) Image-based atmospheric corrections revisited and improved. Photogrammetric Engineering and Remote Sensing 62(9):1025–1036

Chen Q, Han R, Ye F, Li W (2011) Spatio-temporal ecological models. Ecological Informatics 6(1):37 – 43

Chen TS, Lin HJ (2013) Development of a framework for landscape assessment of Taiwanese wetlands. Ecological Indicators 25:121 – 132

Chen XL, Zhao HM, Li PX, Yin ZY (2006) Remote sensing image-based analysis of the relationship between urban heat island and land use/cover changes. Remote Sensing of Environment 104(2):133–146

Chust G, Ducrot D (2004) Land cover mapping with patch-derived landscape indices. Landscape and Urban Planning 69:437–449

Chuvieco E (2002) Teledetección ambiental. La observación de la tierra desde el espacio, 1st edn. Editorial Ariel, Barcelona

Clarke KR (1993) Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology 18(1):117–143

Conesa C (1990) El Campo de Cartagena: clima e hidrología en un medio semiárido, Cuadernos, vol 13, 1st edn. Universidad de Murcia, Murcia

Conesa HM, Jiménez-Cárceles FJ (2007) The Mar Menor lagoon (SE Spain): A singular natural ecosystem threatened by human activities. Marine pollution bulletin 54(7):839–849

Congalton RG (1991) A review of assessing the accuracy of classifications of remotely sensed data. Remote Sensing of Environment 37(1):35–46

Cools J, Johnston R, Hattermann FF, Douven W, Zsuffa I (2013) Tools for wetland management: Lessons learnt from a comparative assessment. Environmental Science & Policy (0):–

COP 10 Decision X/2 (2010) Strategic Plan for Biodiversity 2011 - 2020 and the Aichi Target – Living in Harmony: http://www.cbd.int/sp/targets/

Costanza R, Voinov A (2001) Modeling ecological and economic systems with STELLA: Part III. Ecological Modelling 143(1 - 2):1 – 7

Costanza R, Voinov A (2004) Landscape Simulation Modeling:: A Spatially Explicit, Dynamic Approach. Springer, New York

Council of Europe (1992) Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora

Crawley MJ (2007) The R book. Wiley-Blackwell, Chichester

Cronk JK, Fennessy MS (2001) Wetland plants: biology and ecology. CRC, New York

Dale VH, Beyeler SC (2001) Challenges in the development and use of ecological indicators. Ecological Indicators 1:3–10

Damgaard C (2003) Modeling plant competition along an environmental gradient. Ecological Modelling 170(1):45 – 53

Dauvin JC (2007) Paradox of estuarine quality: Benthic indicators and indices, consensus or debate for the future. Marine Pollution Bulletin 55(1 - 6):271 – 281

Development Team (2009) Quantum GIS Geographic Information System. Open Source Geospatial Foundation Project. Http://www.qgis.org

Development Team (2010) SAGA GIS: System for Automated Geoscientific Analyses. Http://www.saga-gis.org/en/index.html

DGPNB (2008) Consejería de Agricultura y Agua. Comunidad Autónoma de la Región de Murcia. NATMUR-08 project. http://www.murcianatural.com/natmur08/

Drusch M, Bello UD, Carlier S, Colin O, Fernandez V, Gascon F, Hoersch B, Isola C, Laberinti P, Martimort P, Meygret A, Spoto F, Sy O, Marchese F, Bargellini P (2012) Sentinel-2: ESA's Optical High-Resolution Mission for GMES Operational Services. Remote Sensing of Environment 120:25 – 36

Dubois G, Schulz M, Skøien J, Bastin L, Peedell S (2013) eHabitat, a multi-purpose Web Processing Service for ecological modeling. Environmental Modelling & Software 41(0):123 – 133

Dudgeon D, Arthington AH, Gessner MO, Kawabata ZI, Knowler DJ, Lévêque C, Naiman RJ, Prieur-Richard AH, Soto D, Stiassny MLJ, Sullivan CA (2006) Freshwater biodiversity: importance, threats, status and conservation challenges. Biological Reviews 81(2):163–182

Duro DC, Franklin SE, Dubé MG (2012) A comparison of pixel-based and object-based image analysis with selected machine learning algorithms for the classification of agricultural landscapes using SPOT-5 HRG imagery. Remote Sensing of Environment 118:259–272

Engloner AI (2009) Structure, growth dynamics and biomass of reed (*Phragmites australis*) – A review. Flora pp 1–16

Esteve MA (2003) Los humedales de la Región de Murcia: recopilación de valores faunísticos asociados a humedales de zonas áridas. Instituto de Ciencias Sociales y Ambientales (ICSA), Murcia

Esteve MA, Caballero JM, Giménez A (1995) Los paisajes del agua en la Región de Murcia: caracterización ambiental y perspectivas de gestión de los humedales. In: Agua y Futuro en la Región de Murcia, pp 301–341

Esteve MA, Robledano F, Anadón JD, Giménez A (2006) Los ecosistemas de la Región de Murcia: componentes, estructura y dinámica. In: El medio físico de la Región de Murcia, EDITUM. Universidad de Murcia, Murcia, pp 245–278

Esteve MA, Carreño MF, Robledano F, Martínez-Fernández J, Miñano J (2008) Dynamics of coastal wetlands and land use changes in the watershed: Implications for the biodiversity. In: Wetlands: Ecology, Conservation and Restoration., Nova Science Publishers Inc, Hauppauge New York, chap 4, pp 133–175

Esteve MA, Carreño MF, Robledano F, Martínez J, Martínez Paz JM, Miñano J (2012) Protección de humedales y dinámica de la cuenca en la comarca del Mar Menor. In: Los espacios naturales protegidos de la Región de Murcia: estudio de casos desde una perspectiva interdisciplinar, Editum, pp 181–214

European Commission (2000) Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Romania

European Commission (2003) Horizontal guidance document on the role of wetlands in the water framework directive. Common Implementation Strategy for the Water Framework Directive (2000/60/EC)

European Commission – Directorate General for Environment (2007) Interpretation manual of European Union habitats

Fancy S, Gross J, Carter S (2009) Monitoring the condition of natural resources in US national parks. Environmental Monitoring and Assessment 151(1-4):161–174

Felicísimo AM (1994) Modelos digitales del terreno. Pentalfa, Oviedo

Ferreira J, Vale C, Soares C, Salas F, Stacey P, Bricker S, Silva M, Marques J (2007) Monitoring of coastal and transitional waters under the E.U. Water Framework Directive. Environmental Monitoring and Assessment 135:195–216

Finlayson C (2005) Ecosystems & human well-being: wetlands & water synthesis. World Resources Institute, Washington

Fitz HC, Kiker GA, Kim JB (2011) Integrated Ecological Modeling and Decision Analysis Within the Everglades Landscape. Critical Reviews in Environmental Science and Technology 41(sup1):517–547

Foody GM (2002) Status of land cover classification accuracy assessment. Remote Sensing of Environment 80(1):185–201

Franklin SE, Wulder MA (2002) Remote sensing methods in medium spatial resolution satellite data land cover classification of large areas. Progress in Physical Geography 26(2):173–205

Friess D, Spencer T, Smith G, Möller I, Brooks S, Thomson A (2012) Remote sensing of geomorphological and ecological change in response to saltmarsh managed realignment, The Wash, UK. International Journal of Applied Earth Observation and Geoinformation 18:57 – 68

García P, Zapico E, Colubi A (2009) An angiosperm quality index (AQI) for Cantabrian estuaries. Ecological Indicators 9(5):856 – 865

García-García V (2013) Mediterranean wetland functionality in agricultural landscapes. Driving factors for spatial and temporal variability in N and P retention. PhD thesis, Universidad de Murcia

García-García V, Gómez R, Vidal-Abarca MR, Suárez ML (2009) Nitrogen retention in natural Mediterranean wetland-streams affected by agricultural runoff. Hydrology and Earth System Sciences 13(12):2359–2371

García-García V, Gómez R, Vidal-Abarca MR, Suárez ML (2013) Subsurface N retention in two Mediterranean wetland-streams affected by agricultural runoff. Wetlands 33(4):597–608

García Peiró I, Robledano-Aymerich F, Selma E, Ángel M (2009) Abundancias y densidades relativas de zorro *Vulpes vulpes* (Linnaeus, 1758) en un humedal del Sudeste Ibérico: implicaciones para la conservación de sus poblaciones. Anales de Biología 31:43–48

Gardner RH, Engelhardt KA (2008) Spatial processes that maintain biodiversity in plant communities. Perspectives in Plant Ecology, Evolution and Systematics 9(3–4):211 – 228

GBIF (2013) Global Biodiversity Information Facility. Http://www.gbif.org

Gebbert S, Pebesma E (2014) TGRASS: A temporal GIS for field based environmental modeling. Environmental Modelling & Software 53:1 – 12

Gergel SE, Turner MG, Miller JR, Melack JM, Stanley EH (2002) Landscape indicators of human impacts to riverine systems. Aquatic Sciences 64:118–128

Giordani G, Zaldívar J, Viaroli P (2009) Simple tools for assessing water quality and trophic status in transitional water ecosystems. Ecological Indicators 9(5):982 – 991

Gislason PO, Benediktsson JA, Sveinsson JR (2006) Random forests for land cover classification. Pattern Recognition Letters 27(4):294 – 300

Glenn EP, Huete AR, Nagler PL, Nelson SG (2008) Relationship between remotely-sensed vegetation indices, canopy attributes and plant physiological processes: What vegetation indices can and cannot tell us about the landscape. Sensors 8(4):2136–2160

Gómez R (1995) Función de los humedales en la dinámica de nutrientes (N y P) de una cuenca de características áridas: experiencias en el Sureste Ibérico. PhD thesis, Universidad de Murcia

Gómez R, Vidal-Abarca MR, Suárez ML, Fisher SG (1995) The spatial and temporal scale problem in arid zones wetlands management. In: Bases ecológicas para la restauración de humedales en la Cuenca Mediterránea, Junta de Andalucía, Sevilla

Gómez R, Moreno J, Pérez B, Vidal-Abarca MR, Suárez ML (1997) Valores naturales y potencialidades de uso de los humedales asociados a ramblas del Sureste Ibérico. Departamento de Ecología e Hidrología Facultad de Biología Universidad de Murcia

González JC (2011) Propuesta de un método eficiente y de bajo coste para la elaboración de mapas de cultivos mediante imágenes Landsat. El caso de los cítiricos en la Región de Murcia (España). PhD thesis, Universidad de Murcia

González-Alcaraz M, Jiménez-Cárceles F, Álvarez Y, Rogel JA (2013a) Gradients of soil salinity and moisture, and plant distribution, in a Mediterranean semiarid saline watershed: a model of soil-plant relationships for contributing to the management. CATENA DOI:10.1016/j.catena.2013.11.011

González-Alcaraz MN, Conesa HM, Rogel JA (2013b) Phytomanagement of strongly acidic, saline eutrophic wetlands polluted by mine wastes: The influence of liming and *Sarcocornia fruticosa* on metals mobility. Chemosphere 90(10):2512 – 2519

Goslee SC (2011) Analyzing Remote Sensing Data in R: The landsat Package. Journal of Statistical Software 43(4):1–25

GRASS Development Team (2008) Geographic resources analysis support system software. Http://grass.fbk.eu/

Harding JS, Benfield EF, Bolstad PV, Helfman GS, Jones EBD (1998) Stream biodiversity: The ghost of land use past. Proceedings of the National Academy of Sciences 95(25):14,843–14,847

Hattermann F, Krysanova V, Habeck A, Bronstert A (2006) Integrating wetlands and riparian zones in river basin modelling. Ecological Modelling 199(4):379 – 392

Hay GJ, Castilla G, Wulder MA, Ruiz JR (2005) An automated object-based approach for the multiscale image segmentation of forest scenes. International Journal of Applied Earth Observation and Geoinformation 7(4):339 – 359

Herrero J, Castañeda C (2009) Delineation and functional status monitoring in small saline wetlands of NE Spain. Journal of Environmental Management 90(7):2212 – 2218

Herrero J, Snyder R (1997) Aridity and irrigation in Aragon, Spain. Journal of Arid Environments 35(3):535 – 547

Hijmans RJ (2013) raster: Geographic data analysis and modeling. R package version 2.1-49

Hijmans RJ, Phillips S, Leathwick J, Elith J (2013) dismo: Species distribution modeling. R package version 0.8-17

Hirata Y, Takahashi T (2011) Image segmentation and classification of Landsat Thematic Mapper data using a sampling approach for forest cover assessment. Canadian Journal of Forest Research 41(1):35–43

Hoch R, Gabele T, Benz J (1998) Towards a standard for documentation of mathematical models in ecology. Ecological Modelling 113(1–3):3 – 12

Hollenhorst TP, Brown TN, Johnson LB, Ciborowski JJH, Host GE (2007) Methods for generating multi-scale watershed delineations for indicator development in great lake coastal ecosystems. Journal of Great Lakes Research 33(3):13–26

Hollis G (1990) Environmental impacts of development on wetlands in arid and semi-arid lands/les effets sur l'environnement de la mise en valeur des zones humides dans ses regions arides et semi arides. Hydrological Sciences Journal 35(4):411–428

Hollis GE, Thompson JR (1998) Hydrological data for wetland management. Water and Environment Journal 12(1):9–17

Horning N, Robinson JA, Sterling EJ, Turner W, Spector S (2010) Remote Sensing for Ecology and Conservation: A Handbook of Techniques. Oxford Univ Press

Houlahan JE, Findlay CS (2004) Estimating the 'critical' distance at which adjacent land-use degrades wetland water and sediment quality. Landscape Ecology 19(6):677–690

Huete A (1988) A soil-adjusted vegetation index (SAVI). Remote Sensing of Environment 25(3):295 – 309

Hughes RG (2004) Climate change and loss of saltmarshes: consequences for birds. Ibis 146:21–28

Hui F, Xu B, Huang H, Yu Q, Gong P (2008) Modelling spatial-temporal change of Poyang Lake using multitemporal Landsat imagery. International Journal of Remote Sensing 29(19-20):18

Husson F, Josse J, Le S, J M (2010) FactoMineR: Multivariate Exploratory Data Analysis and Data Mining with R. R package version 1.14

Innis SA, Naiman RJ, Elliott SR (2000) Indicators and assessment methods for measuring the ecological integrity of semi-aquatic terrestrial environments. Hydrobiologia 422-423:111–131

Irons JR, Dwyer JL, Barsi JA (2012) The next Landsat satellite: The Landsat Data Continuity Mission. Remote Sensing of Environment 122:11 – 21

ISEE Systems Inc (2013) Stella and iThink software. Http://www.iseesystems.com

Jachner S, van den Boogaart G, Petzoldt T (2007) Statistical methods for the qualitative assessment of dynamic models with time delay (R Package qualV). Journal of Statistical Software 22(9):1–31

Jakeman A, Letcher R, Norton J (2006) Ten iterative steps in development and evaluation of environmental models. Environmental Modelling & Software 21(5):602 – 614

Jenkins WA, Murray BC, Kramer RA, Faulkner SP (2010) Valuing ecosystem services from wetlands restoration in the Mississippi Alluvial Valley. Ecological Economics 69(5):1051 – 1061

Ji W (2008) Wetland and water resource modeling and assessment : a watershed perspective. CRC Press, Boca Raton

Johnston CA, Zedler JB, Tulbure MG, Frieswyk CB, Bedford BL, Vaccaro L (2009) A unifying approach for evaluating the condition of wetland plant communities and identifying related stressors. Ecological Applications 19(7):1739–1757

Jones K, Lanthier Y, van der Voet P, van Valkengoed E, Taylor D, Fernández-Prieto D (2009) Monitoring and assessment of wetlands using Earth Observation: The GlobWetland project. Journal of Environmental Management 90(7):2154–2169

Jørgensen SE (2008) Overview of the model types available for development of ecological models. Ecological Modelling 215(1 - 3):3 – 9

Jørgensen SE, Bendoricchio G (2001) Fundamentals of ecological modelling. Elsevier, Amsterdam

Karssenberg D, de Jong K, van der Kwast J (2007) Modelling landscape dynamics with python. International Journal of Geographical Information Science 21(5):483–495

King RS, Baker ME, Whigham DF, Weller DE, Jordan TE, Kazyak PF, Hurd MK (2005) Spatial considerations for linking watershed land cover to ecological indicators in streams. Ecological Applications 15(1):137–153

Klemas V (2001) Remote sensing of landscape-level coastal environmental indicators. Environmental Management 27:47–57

Klemas V (2011) Remote sensing of wetlands: case studies comparing practical techniques. Journal of Coastal Research 27(3):418–427

Lee TM, Yeh HC (2009) Applying remote sensing techniques to monitor shifting wetland vegetation: A case study of Danshui River estuary mangrove communities, Taiwan. Ecological Engineering 35(4):487–496

Levin N, Elron E, Gasith A (2009) Decline of wetland ecosystems in the coastal plain of Israel during the 20th century: Implications for wetland conservation and management. Landscape and Urban Planning 92(3-4):220–232

Liaw A, Wiener M (2002) Classification and regression by randomforest. R News 2(3):18–22

Logan M (2010) Biostatistical design and analysis using R a practical guide. Wiley-Blackwell, Chichester, UK; Hoboken, NJ

López RD, Fennessy MS (2002) Testing the floristic quality assessment index as an indicator of wetland condition. Ecological Applications 12(2):487–497

Lu D, Weng Q (2007) A survey of image classification methods and techniques for improving classification performance. International Journal of Remote Sensing 28(5):823–870

Lucena-Moya P, Pardo I (2012) An invertebrate multimetric index to classify the ecological status of small coastal lagoons in the Mediterranean ecoregion (MIBIIN). Mar Freshwater Res 63(9):801–814

Mack J (2006) Landscape as a predictor of wetland condition: an evaluation of the Landscape Development Index (LDI) with a large reference wetland dataset from Ohio. Environmental Monitoring and Assessment 120:221–241

MacKay H, Finlayson CM, Fernández-Prieto D, Davidson N, Pritchard D, Rebelo LM (2009) The role of Earth Observation (EO) technologies in supporting implementation of the Ramsar Convention on Wetlands. Journal of Environmental Management 90(7):2234–2242

Magnussen S, Boudewyn P, Wulder M (2004) Contextual classification of Landsat TM images to forest inventory cover types. International Journal of Remote Sensing 25(12):2421–2440

Maheu-giroux M, Blois SD (2005) Mapping the invasive species *Phragmites australis* in linear wetland corridors. Aquatic Botany 83:310–320

Maltby E, Acreman MC (2011) Ecosystem services of wetlands: pathfinder for a new paradigm. Hydrological Sciences Journal 56(8):1341–1359

Maltby E, Acreman M, Blackwell M, Everard M, Morris J (2013) The challenges and implications of linking wetland science to policy in agricultural landscapes – experience from the UK National Ecosystem Assessment. Ecological Engineering 56(0):121 – 133

MARM (2009) Bases ecológicas preliminares para la conservación de los tipos de hábitat de interés comunitario presentes en España. Madrid

Martín-Queller E, Moreno-Mateos D, Pedrocchi C, Cervantes J, Martínez G (2010) Impacts of intensive agricultural irrigation and livestock farming on a semi-arid Mediterranean catchment. Environmental Monitoring and Assessment 167:423–435

Martínez-Fernández J, Esteve-Selma MA, Robledano-Aymerich F, Pardo MT, Carreño-Fructuoso MF (2005) Aquatic birds as bioindicators of trophic changes and ecosystem deterioration in the Mar Menor lagoon (SE Spain). Hydrobiologia 550:221–235

Martínez-López J, Carreño MF, Palazón-Ferrando JA, Martínez-Fernández J, Esteve-Selma MA (2012) Wetland-Watershed Modelling and Assessment: GIS Methods for Establishing Multiscale Indicators. In: Wetlands: Ecology, Management and Conservation, Nova Science Publishers Inc, Hauppauge New York, pp 231–250

Martínez-López J, Carreño MF, Palazón-Ferrando JA, Martínez-Fernández J, Esteve MA (2013) Free advanced modeling and remote-sensing techniques for wetland watershed delineation and monitoring. International Journal of Geographical Information Science pp 1–16, DOI:10.1080/13658816.2013.852677

Martínez-López J, Carreño MF, Martínez-Fernández J, Esteve MA (2014a) Wetland and landscape indices for assessing the condition of semiarid Mediterranean saline wetlands under agricultural hydrological pressures. Ecological Indicators 36(C):400 – 408

Martínez-López J, Carreño MF, Palazón-Ferrando JA, Martínez-Fernández J, Esteve MA (2014b) Remote sensing of plant communities as a tool for assessing the condition of semiarid Mediterranean saline wetlands in agricultural catchments. International Journal of Applied Earth Observation and Geoinformation 26(0):193 – 204

Maxwell T, Costanza R (1997) A language for modular spatio-temporal simulation. Ecological Modelling 103(2 - 3):105 – 113

McHugh OV, McHugh AN, Eloundou-Enyegue PM, Steenhuis TS (2007) Integrated qualitative assessment of wetland hydrological and land cover changes in a data scarce dry Ethiopian highland watershed. Land Degradation & Development 18(6):643–658

Michelson DB, Liljeberg BM, Pilesjö P (2000) Comparison of algorithms for classifying Swedish landcover using Landsat TM and ERS-1 SAR data. Remote Sensing of Environment 71(1):1–15

Millán A, Abellán P, Sánchez-Fernández D, Gutiérrez-Cánovas C, Picazo F, Arribas P, Belmar O, Velasco J (2009) Biodiversidad de macroinvertebrados en los ecosistemas acuáticos salinos en la región de murcia. Los Paisajes Ibéricos de la Sal 2:111–126

Miller SJ, Wardrop DH, Mahaney WM, Brooks RP (2006) A plant-based index of biological integrity (IBI) for headwater wetlands in central Pennsylvania. Ecological Indicators 6:290–312

Mitsch W (2013) Carbon, wetlands and climate change: a global perspective. In: ISEM Conference - Ecological Modelling for Ecosystem Sustainability in the context of Global Change, Toulouse (France)

Mitsch WJ, Gosselink JG (2007) Wetlands, 4th edn. Wiley, Hoboken, N.J

Moffett K, Robinson D, Gorelick S (2010) Relationship of Salt Marsh Vegetation Zonation to Spatial Patterns in Soil Moisture, Salinity, and Topography. Ecosystems pp 1–16

Moloney KA, Jeltsch F (2008) Space matters - Novel developments in plant ecology through spatial modelling. Perspectives in Plant Ecology, Evolution and Systematics 9(3 - 4):119 – 120

Moreno JL, Vidal-Abarca MR, Suárez ML (2001) Ecosistemas acuáticos de las ramblas murcianas: ecología y conservación de sus arroyos y humedales asociados. Quercus (184):28–33

Muetzelfeldt R, Massheder J (2003) The simile visual modelling environment. European Journal of Agronomy 18(3–4):345 – 358

Muxika I, Borja A, Bonne W (2005) The suitability of the marine biotic index (AMBI) to new impact sources along European coasts. Ecological Indicators 5(1):19 – 31

Muxika I, Borja A, Bald J (2007) Using historical data, expert judgement and multivariate analysis in assessing reference conditions and benthic ecological status, according to the European Water Framework Directive. Marine Pollution Bulletin 55(1 - 6):16 – 29

Muxika I, Somerfield PJ, Borja A, Warwick RM (2012) Assessing proposed modifications to the AZTI marine biotic index (AMBI), using biomass and production. Ecological Indicators 12(1):96 – 104

Mwita E, Menz G, Misana S, Becker M, Kisanga D, Boehme B (2013) Mapping small wetlands of Kenya and Tanzania using remote sensing techniques. International Journal of Applied Earth Observation and Geoinformation 21:173 – 183

Naimi B, Voinov A (2012) StellaR: A software to translate Stella models into R open-source environment. Environmental Modelling & Software 38(0):117 – 118

Narumalani S, Zhou Y, Jelinski DE (1998) Utilizing geometric attributes of spatial information to improve digital image classification. Remote Sensing Reviews 16(4):233–253

Neteler M, Bowman MH, Landa M, Metz M (2012) GRASS GIS: A multi-purpose open source GIS. Environmental Modelling & Software 31(0):124 – 130

Newton AC, Hill RA, Echeverría C, Golicher D, Rey Benayas JM, Cayuela L, Hinsley SA (2009) Remote sensing and the future of landscape ecology. Progress in Physical Geography 33(4):528–546

Niemeijer D, Groot RSD (2008) A conceptual framework for selecting environmental indicator sets. Ecological indicators 8:14–25

Niemi GJ, McDonald ME (2004) Application of ecological indicators. Annual Review of Ecology, Evolution, and Systematics 35:89–111

O'Callaghan JF, Mark DM (1984) The extraction of drainage networks from digital elevation data. Computer Vision, Graphics, and Image Processing 28(3):323 – 344

O'Connell M (2003) Detecting, measuring and reversing changes to wetlands. Wetlands Ecology and Management 11:397–401

O'Connor JS, Dewling RT (1986) Indices of marine degradation: Their utility. Environmental Management 10(3):335–343

Okruszko T, Duel H, Acreman M, Grygoruk M, Floerke M, Schneider C (2011) Broad-scale ecosystem services of European wetlands—overview of the current situation and future perspectives under different climate and water management scenarios. Hydrological Sciences Journal 56(8):1501–1517, DOI 10.1080/02626667.2011.631188

Oksanen J, Blanchet G, Kindt R, Legendre P, O'Hara RB, Simpson G, Solymos P, S, Wagner H (2011) vegan: Community Ecology Package. R package version 1.17-8

Olaya Ferrero V (2004) Hidrología Computacional y Modelos Digitales de Terreno: Teoría, práctica y filosofía de una nueva forma de análisis hidrológico. URL http://www.gabrielortiz.com/descargas/Hidrologia_Computacional_MDT_SIG.pdf

Ornat AL, Morales C (2002) Integrated Water Management to Address Environmental Degradation in the Mediterranean Region. IUCN Mediterranean Office

Ortega M, Velasco J, Millan A, Guerrero C (2004) An ecological integrity index for littoral wetlands in agricultural catchments of semiarid Mediterranean regions. Environmental Management 33(3):412–430

Ortiz R, Álvarez Rogel J, Alcaraz F (1995) Soil-vegetation relationships in 2 coastal salt marshes in Southeastern Spain. Arid Soil Research and Rehabilitation 9(4):481–493

Ozesmi S, Bauer M (2002) Satellite remote sensing of wetlands. Wetlands Ecology and Management 10(5):381–402

Pal M (2005) Random forest classifier for remote sensing classification. International Journal of Remote Sensing 26(1):217–222

Pardo M (2002) Relaciones Especies-Área en los Saladares del Guadalentín (SE Ibérico). Tésis de Licenciatura, Universidad de Murcia

Pardo MT, Esteve MA, Giménez A, Martínez-Fernández J, Carreño MF, Serrano J, Miñano J (2008) Assessment of hydrological alterations on wandering beetle assemblages (coleoptera: Carabidae and Tenebrionidae) in coastal wetlands of arid Mediterranean systems. Journal of Arid Environments 72(10):1803–1810

Pat S Chavez J (1988) An improved dark-object subtraction technique for atmospheric scattering correction of multispectral data. Remote Sensing of Environment 24(3):459–479

Pebesma E (2012) spacetime: Spatio-Temporal Data in R. Journal of Statistical Software 51(7)

Pennings SC, Callaway RM (1992) Salt marsh plant zonation: the relative importance of competition and physical factors. Ecology pp 681–690

Perry GL, Millington JD (2008) Spatial modelling of succession-disturbance dynamics in forest ecosystems: Concepts and examples. Perspectives in Plant Ecology, Evolution and Systematics 9(3–4):191 – 210

Petzoldt T, Rinke K (2007) simecol: An Object-Oriented Framework for Ecological Modeling in R. Journal of Statistical Software 22(9):1–31

Poisot T, Mounce R, Gravel D (2013) Moving toward a sustainable ecological science: don't let data go to waste! Ideas in Ecology and Evolution 6(2)

Poulin B, Davranche A, Lefebvre G (2010) Ecological assessment of *Phragmites australis* wetlands using multi-season SPOT-5 scenes. Remote Sensing of Environment 114(7):1602–1609

Pujol Fructuoso JA (2002) Ecología de la germinación de semillas y del crecimiento del plántulas de especies halófitas del SE Ibérico. Instituto Municipal de Cultura Joaquín Chapaprieta, Alicante

Pullar D (2004) Simumap: a computational system for spatial modelling. Environmental Modelling & Software 19(3):235 – 243

Qi J, Chehbouni A, Huete A, Kerr Y, Sorooshian S (1994) A modified soil adjusted vegetation index. Remote Sensing of Environment 48(2):119 – 126

R Core Team (2013) R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria

Ramírez-Díaz L, Calvo JF, Enríquez JL, Esteve MA, Giménez A, Gómez R, González G, Hernández V, Millán A, Nicolás E (1992) Los humedales de la Región de Murcia: Tipificación, Cartografía y Plan de Gestión para la Conservación. Caja de Ahorros del Mediterráneo/Comunidad Autónoma de la Región de Murcia

Ramírez-Díaz L, Vidal-Abarca MR, Calvo JF, Suarez ML, Palazón JA, Esteve MA, Gómez R, Jimenez A, Pujol JA, Sánchez JA (1995) Bases ecológicas para la delimitación, ordenación y gestión del Paisaje Protegido de las Ramblas de Ajauque y Salada. Universidad de Murcia, Murcia

Ramsar Convention Secretariat (2004) The Ramsar Handbook for the Wise Use of Wetlands. Gland

Richards JA, Jia X (2006) Remote Sensing Digital Image Analysis: An Introduction, 4th edn. Springer, Heidelberg

Ripley BD (2001) The R Project in Statistical Computing. MSOR Connections 1(1):23–25

Roberts D (2010) labdsv: Ordination and Multivariate Analysis for Ecology. R package version 1.4-1

Robledano-Aymerich F, Calvo JF, Esteve Selma MA, Mas Hernández J, Palazón JA, Suárez ML, Torres A, Vidal-Abarca MR, Ramírez-Díaz L (1991a) Estudios ecológicos de los humedales costeros del Sudeste español. I: inventario y tipificación. Anales de Biología (17):153–163

Robledano-Aymerich F, Calvo Sendín JF, Esteve Selma MA, Mas Hernández J, Palazón JA, Suárez ML, Torres Martínez A, Vidal-Abarca MR, Ramírez-Díaz L (1991b) Estudios ecológicos de los humedales costeros del Sudeste español. II: evolución histórica, situación actual y perspectivas de conservación. Anales de Biología (17):165–176

Robledano-Aymerich F, Montes del Olmo C, Ramírez Díaz L (1992) Relaciones ambientales y conservación de las comunidades de aves acuáticas en la gestión de los humedales del Sudeste español, vol 33. EDITUM

Robledano-Aymerich F, Esteve MA, Farinós P, Carreño MF, Martínez-Fernández J (2010) Terrestrial birds as indicators of agricultural-induced changes and associated loss in conservation value of Mediterranean wetlands. Ecological Indicators 10(2):274–286

Rocchini D, Neteler M (2012) Let the four freedoms paradigm apply to ecology. Trends in Ecology & Evolution 27(6):310–311

Rooney R, Bayley S, Creed I, Wilson M (2012) The accuracy of land cover-based wetland assessments is influenced by landscape extent. Landscape Ecology pp 1–15

Roth NE, Allan JD, Erickson DL (1996) Landscape influences on stream biotic integrity assessed at multiple spatial scales. Landscape Ecology 11(3):141–156

Rouse JW, Haas RH, Schell JA, Deering DW (1973) Monitoring vegetation systems in the Great Plains with ERTS. In: Proceedings of the Third ERTS Symposium, NASA, vol 1, pp 309–317

RStudio Inc (2013) shiny: Web Application Framework for R. R package version 0.7.0

Sanchez JM, Otero XL, Izco J (1998) Relationships between vegetation and environmental characteristics in a salt-marsh system on the coast of Northwest Spain. Plant Ecology 136(1):1–8

Schmitz O, Karssenberg D, de Jong K, de Kok JL, de Jong SM (2013) Map algebra and model algebra for integrated model building. Environmental Modelling & Software 48(0):113 – 128

Schwartz S, Jenkins D (2000) Temporary aquatic habitats: constraints and opportunities. Aquatic Ecology 34:3–8

Scikit-image Team (2013) Scikit-image 0.8.0. Http://scikit-image.org/

de Senna Carneiro TG, de Andrade PR, Câmara G, Monteiro AMV, Pereira RR (2013) An extensible toolbox for modeling nature–society interactions. Environmental Modelling & Software 46(0):104 – 117

Serra P, Pons X, Saurí D (2008) Land-cover and land-use change in a Mediterranean landscape: A spatial analysis of driving forces integrating biophysical and human factors. Applied Geography 28(3):189 – 209

Sluiter R (2005) Mediterranean land cover change: modelling and monitoring natural vegetation using GIS and remote sensing. Netherlands Geographical Studies 333:17–144

Smith GM, Fuller RM (2001) An integrated approach to land cover classification: An example in the Island of Jersey. International Journal of Remote Sensing 22(16):3123–3142

Soetaert K, Petzoldt T, Setzer RW (2010) Solving Differential Equations in R: Package deSolve. Journal of Statistical Software 33(9):1–25

Steiniger S, Bocher E (2009) An overview on current free and open source desktop GIS developments. International Journal of Geographical Information Science 23(10):1345–1370

Steiniger S, Hay GJ (2009) Free and open source geographic information tools for landscape ecology. Ecological Informatics 4(4):183 – 195

Strayer DL, Dudgeon D (2010) Freshwater biodiversity conservation: recent progress and future challenges. Journal of the North American Benthological Society 29(1):344–358

Strayer DL, Beighley RE, Thompson LC, Brooks S, Nilsson C, Pinay G, Naiman RJ (2003) Effects of land cover on stream ecosystems: Roles of empirical models and scaling issues. Ecosystems 6(5):407–423

Stuckens J, Coppin P, Bauer M (2000) Integrating contextual information with per-pixel classification for improved land cover classification. Remote Sensing of Environment 71(3):282 – 296

Suárez ML, Vidal-Abarca MR, Calvo JF, Palazón JA, Esteve MA, Gómez R, Giménez A, Pujol JA, Sánchez JA, Pardo M (1996) Zone humide d'ajauque-rambla salada, espagne. Management of Mediterranean Wetlands 3:39–55

Szantoi Z, Escobedo F, Abd-Elrahman A, Smith S, Pearlstine L (2013) Analyzing fine-scale wetland composition using high resolution imagery and texture features. International Journal of Applied Earth Observation and Geoinformation 23:204 – 212

Tarboton DG (1997) A new method for the determination of flow directions and upslope areas in grid digital elevation models. Water Resources Research 33(2):309–319

Tong STY, Chen W (2002) Modeling the relationship between land use and surface water quality. Journal of Environmental Management 66(4):377–393

Tufto J, Cavallini P (2005) Should wildlife biologists use free software? Wildlife Biology 11(1):67–76

Tulbure MG, Johnston CA, Auger DL (2007) Rapid Invasion of a Great Lakes Coastal Wetland by Non-native *Phragmites australis* and *Typha*. Journal of Great Lakes Research 33, Supplement 3(0):269 – 279

Turner MG, Arthaud GJ, Engstrom RT, Hejl SJ, Liu J, Loeb S, McKelvey K (1995) Usefulness of spatially explicit population models in land management. Ecological Applications 5(1):12–16

Turner R, van den Bergh JC, Soderqvist T, Barendregt A, van der Straaten J, Maltby E, van Ierland EC (2000) Ecological-economic analysis of wetlands: scientific integration for management and policy. Ecological Economics 35(1):7 – 23

Turner RK, Georgiou S, Brouwer R, Bateman IJ, Langford IJ (2003) Towards an integrated environmental assessment for wetland and catchment management. The Geographical Journal 169(2):99–116

Černá L, Chytrý M (2005) Supervised classification of plant communities with artificial neural networks. Journal of Vegetation Science 16(4):407–414

Van Sickle J, Johnson CB (2008) Parametric distance weighting of landscape influence on streams. Landscape Ecology 23(4):427–438

Velasco J, Lloret J, Millan A, Marin A, Barahona J, Abellan P, Sanchez-Fernandez D (2006) Nutrient and particulate inputs into the Mar Menor lagoon (SE Spain) from an intensive agricultural watershed. Water, Air, and Soil Pollution 176(1-4):37–56

Vicente MJ, Conesa E, Rogel JA, Franco JA, Martínez-Sánchez JJ (2007) Effects of various salts on the germination of three perennial salt marsh species. Aquatic Botany 87(2):167 – 170

Vidal-Abarca MR, Esteve MA, Suárez ML, Gómez R, Robledano F, Martinez J, Martínez B (1998) Análisis de viabilidad del uso de humedales para mitigar el efecto de los vertidos y drenajes agrícolas al Mar Menor. Informe técnico para la Confederación Hidrográfica del Segura, Murcia

Vidal-Abarca MR, Suárez ML, Ramírez-Díaz L (2000a) Tipo funcional de humedal: ramblas/wadis. Management of Mediterranean Wetlands 3:17–38

Vidal-Abarca MR, Suárez Alonso ML, Gómez Cerezo RM (2000b) Los humedales: ecosistemas para conservar. In: Biodiversidad: contribución a su conocimiento y conservación en la región de Murcia, Servicio de Publicaciones, pp 149–162

Vidal-Abarca MR, Esteve MA, Suárez ML, Gómez R, Robledano F (2003) Los humedales de la Región de Murcia: Claves para su interpretación. Consejería de Agricultura, Agua y Medio Ambiente. Dirección General del Medio Natural, Murcia

Vieira MA, Formaggio AR, Rennó CD, Atzberger C, Aguiar DA, Mello MP (2012) Object based image analysis and data mining applied to a remotely sensed landsat time-series to map sugarcane over large areas. Remote Sensing of Environment 123(0):553 – 562

Voinov A, Fitz C, Boumans R, Costanza R (2004) Modular ecosystem modeling. Environmental Modelling & Software 19(3):285 – 304

Voinov AA, DeLuca C, Hood RR, Peckham S, Sherwood CR, Syvitski JPM (2010) A community approach to earth systems modeling. Eos, Transactions American Geophysical Union 91(13):117–118

Wang C, Menenti M, Stoll M, Belluco E, Marani M (2007) Mapping mixed vegetation communities in salt marshes using airborne spectral data. Remote Sensing of Environment 107(4):559–570

Wang SH, Huang SL, Budd WW (2012) Integrated ecosystem model for simulating land use allocation. Ecological Modelling 227(0):46 – 55

Watson FG, Rahman JM (2004) Tarsier: a practical software framework for model development, testing and deployment. Environmental Modelling & Software 19(3):245 – 260

Weber J, Uhel R, Spyropoulou R, Breton F, Arévalo J, Richard D, Haines-Young R, Potschin M, Kumar P, Martin B (2010) Ecosystem accounting for the cost of biodiversity losses: framework and case study for coastal Mediterranean wetlands. An EEA–European Environmental Agency study (Phase I), Copenhagen

Whiteside TG, Boggs GS, Maier SW (2011) Comparing object-based and pixel-based classifications for mapping savannas. International Journal of Applied Earth Observation and Geoinformation 13(6):884 – 893

Wickham H, Chang W (2013) devtools: Tools to make developing R code easier. R package version 1.4

Wigand C, Comeleo R, McKinney R, Thursby G, Chintala M, charpentier M (1999) Outline of a new approach to evaluate ecological integrity of salt marshes. Human and Ecological Risk Assessment: An International Journal 5(7):1541–1554

Williams W (1999) Conservation of wetlands in drylands: a key global issue. Aquatic Conservation: Marine and Freshwater Ecosystems 9(6):517–522

Williams W (2002) Environmental threats to salt lakes and the likely status of inland saline ecosystems in 2025. Environmental Conservation 29:154–167

Wilson G, Aruliah DA, Brown CT, Hong NPC, Davis M, Guy RT, Haddock SHD, Huff K, Mitchell I, Plumbley M, Waugh B, White EP, Wilson P (2013) Best Practices for Scientific Computing. http://arxiv.org/abs/1210.0530

Wolter P, Johnston C, Niemi G (2006) Land use land cover change in the US Great Lakes basin 1992 to 2001. Journal of Great Lakes Research 32(3):607–628

Xie Y, Sha Z, Yu M (2008) Remote sensing imagery in vegetation mapping: a review. Journal of Plant Ecology 1(1):9–23

Yan G, Mas J, Maathuis B, Xiangmin Z, Van Dijk P (2006) Comparison of pixel based and object oriented image classification approaches – a case study in a coal fire area, Wuda, Inner Mongolia, China. International Journal of Remote Sensing 27(18):4039–4055

Zedler JB, Kercher S (2005) Wetland resources: Status, trends, ecosystem services, and restorability. Annual Review of Environment and Resources 30(1):39–74

Zedler JB, Callaway JC, Desmond JS, Vivian-Smith G, Williams GD, Sullivan G, Brewster AE, Bradshaw BK (1999) Californian salt-marsh vegetation: An improved model of spatial pattern. Ecosystems 2(1):19–35

Zha Y, Gao J, Ni S (2003) Use of normalized difference built-up index in automatically mapping urban areas from TM imagery. International Journal of Remote Sensing 24(3):583–594

Zhou D, Gong H, Liu Z (2008) Integrated ecological assessment of biophysical wetland habitat in water catchments: Linking hydro-ecological modelling with geo-information techniques. Ecological Modelling 214(2–4):411 – 420

Zorrilla-Miras P, Palomo I, Gómez-Baggethun E, Martín-López B, Lomas P, Montes C (2013) Effects of land-use change on wetland ecosystem services: A case study in the Doñana marshes (SW Spain). Landscape and Urban Planning DOI j.landurbplan.2013.09.013